Sustainable fisheries and

habitat-fishery interactions

Rachel Nichols, B.Ec (Hons), LL.B, Grad.Dip.Legal.Prac

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Statement of Co-Authorship

The following people and institutions contributed to the publication of work undertaken as part of this thesis:

Rachel Nichols (Tasmanian School of Business and Economics) = **Candidate**

Dr. Satoshi Yamazaki (Tasmanian School of Business and Economics) = **Author 1**

Dr. Sarah Jennings (Tasmanian School of Business and Economics) = **Author 2**

Author details and their roles:

Paper 1, The role of precaution in stock recovery plans in a fishery with habitat effect

Located in chapter 2; Candidate (80%), Author 1 (10%), Author 2 (10%).

Candidate was the primary author, with both Author 1 and Author 2 contributing to the interpretation of results and the development of the manuscript.

Paper 2, Allocation of harvest between users in a fishery with habitat effect

Located in chapter 3; Candidate (80%), Author 1 (10%), Author 2 (10%).

Candidate was the primary author, with both Author 1 and Author 2 contributing to the interpretation of results and the development of the manuscript.

We the undersigned agree with the above stated "proportion of work undertaken" for each of the above published (or submitted) peer-reviewed manuscripts contributing to this thesis:

Signed:

Dr. Satoshi Yamazaki

Supervisor

Tasmanian School of Business and Economics

University of Tasmania

Signed:

Dr. Joaquin Vespignani

Acting Head of School

Tasmanian School of Business and Economics

University of Tasmania

Date: 14 December 2018

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Abstract

The sustainable management of fishery resources is a highly complex undertaking, requiring a balance between the socio-economic needs of fishery users and the biological constraints of the fishery resource. The impacts of fishing activity on marine habitats can make this already complex problem even more difficult. This fishing-induced habitat damage may undermine the sustainability of fishery resources by causing irreversible habitat loss, or by slowing the fish stock regeneration process. Given the essential role of fishery resources in providing economic and social benefits for many nations, as well as their vital role in the global food supply, the question of how to sustainably manage these resources where habitat degradation is of concern, is therefore of high importance and interest.

This thesis seeks to improve understanding of how the impacts of fishing activity on marine habitats influence the sustainable management of fishery resources. The objective of this thesis is to explore how information concerning the marine habitat may be incorporated into the decision-making processes of fishery management where fishing-induced habitat degradation is of concern, and the circumstances in which management tools usually used to achieve habitat improvements may generate fishery benefits. This thesis consists of three essays which explore the interplay between fish stocks and the habitats that support these resources, and how this relationship impacts the effectiveness of management mechanisms implemented to optimise across biological and economic outcomes. The first two essays examine how information concerning the habitat may be incorporated into decision-making to improve fishery management outcomes. The first essay examines alternative ways of exercising precaution in stock recovery plans in achieving stock rebuilding when fishing-induced habitat damage occurs, while taking into consideration the economic and socio-economic objectives of fisheries management. The second essay explores the connection between fishing-induced habitat

damage and the optimal allocation of harvest across multiple user groups and explores the consequences of omitting information concerning the marine environment for fishery performance and catch share allocations. The extent of habitat degradation in the first essay is controlled by management through the total protection from fishing mortality, and in the second essay, through the allocation of harvest between user groups which use fishing gears that have different levels of habitat impacts. The third essay undertakes an empirical analysis of the effect of a marine protected area (MPA) network, established for habitat conservation, on adjacent fisheries, using Australia's south-east network of MPAs as a case study. This essay examines the effect of this network on catch and the gross value of production for adjacent fisheries, while considering the potentially confounding influence of fishery management changes occurring in the same time period.

The results in this thesis highlight the importance of having adequate knowledge of the marine habitats in which fishing activity takes place, and of the circumstances in which this knowledge ought to be included when making decisions concerning the management of fishery resources. Results show that the trade-offs between socio-economic outcomes for fishery users and the biological objectives of fishery management may be heightened in the presence of fishing-induced habitat damage. However, the combined use of fishery management tools to control fishing mortality may allow these trade-offs to be avoided, particularly in cases where a comparatively relaxed stock recovery plan is implemented, or where the ecosystem is relatively resilient to habitat damage. Additionally, a failure to incorporate habitat-fishery connections into decisions concerning catch share allocations may have devastating consequences in sensitive environments, due to inadvertent overfishing and the inability of these environments to recover from destructive fishing activity. Finally, results show that, while the implementation of an MPA network for the purposes of habitat management may lead to declines in catch for adjacent fisheries, favourable market conditions may have a compensatory effect, such that overall revenue for adjacent fisheries is not affected. However, the impacts of an MPA network on adjacent fisheries will be highly contextual, and dependent on key factors such as the age of the MPA network, the region-specific effects of climate change, and the ability of fishers to adapt their behaviour in terms of fishing location and target species.

Contents

Chapter 1: Introduction

1.1. Background

Fishery resources are a key source of employment and economic development for coastal communities, as well as forming a vital part of the global food supply. According to the 2018 State of World Fisheries and Aquaculture Report (FAO, 2018), annual global growth in fish consumption has been twice as high as population growth since 1961, with the diets of those in developing nations consisting of a higher share of fish protein than those in developed countries. Currently, 33% of fish stocks globally are considered overfished, with 67% sustainably fished. While the majority of stocks are currently sustainably fished, almost 60% of these are fully exploited (FAO, 2018), which means that only 7% of stocks currently being commercially exploited can accommodate increases in harvest without becoming immediately overfished. As such, fish sourced from wild-capture marine fisheries are not expected to be able to completely meet global demand in the future, unless the sustainable use of fishery resources is promoted (Garcia & Rosenberg, 2010). The importance of the sustainable use of fishery resources has been reflected in global commitments such as the United Nation's sustainable development goals (SDGs), in particular SDG 14, which includes aims such as rebuilding overfished stocks, increasing the number of marine protected areas, and widening the scope of fishery management to include information concerning the broader ecosystem (UN, 2015). Given the pivotal role of these stocks in the global food supply and the reliance of many nations on marine resources as a source of nutrition and economic and social benefits, the question of how to sustainably manage marine resources into the future, and the mechanisms through which this is achieved, is both pertinent and pressing.

The sustainable management of social-ecological systems (SESs) such as fisheries is a highly complex undertaking, requiring consideration of multiple relationships and variables across multiple systems (Ostrom, 2007). SESs are integrated, complex systems which include human and biophysical subsystems in a two-way feedback relationship (Anderies et al., 2004; Berkes, 2011). In the context of fisheries, this relationship may be characterised in terms of the interactions between the ecosystem functions which support the fish populations targeted by the fishery, and the human impacts on the ecosystem generated when targeting those populations (Kittinger et al., 2013). Human impacts on the ecosystem arise not only from the removal of species from that ecosystem, but also from degradation inflicted on marine habitats by fishing activity. The association between fishing activity and habitat damage has been the subject of increased study and concern (Hiddink et al., 2011; Kahui et al., 2016; Shephard et al., 2010; Turner et al., 1999). Habitat degradation may undermine the sustainability of fishery resources by causing irreversible habitat loss, or by slowing the regeneration process of stocks; for example, by negatively impacting reproduction habitats (Sundblad & Bergström, 2014) or inhibiting access to prey (Hiddink et al., 2011). Conversely, improvements to the marine environment may encourage the sustainability of fish stocks, by improving ecosystem resilience through increased biodiversity (Russ & Alcala, 2011) or by increasing the productivity of fish stocks through improved reproduction (Lloret & Planes, 2003). Given these connections between habitat quality and fishery resource abundance, achieving the sustainable management of fishery resources relies on a better understanding of how the human impacts of fishing activity may undermine this management, and how information concerning habitat linkages and processes may be utilised to improve the application of the management tools currently used by fishery managers.

One of the first acknowledgements that sustainable fishery resource management would require a broader approach than had previously been used came with the adoption of the precautionary principle, which has been a cornerstone of natural resource management since its inclusion in the United Nations' World Charter for Nature in 1982. The precautionary principle expresses a desire to prevent damage to the environment before it occurs and, if damage has previously occurred, not to postpone or avoid taking action due to scientific uncertainty. In the context of fishery management, the objective of the precautionary principle is to prevent resource and environmental degradation, while taking into consideration the economic and socio-economic requirements of fisheries (Garcia, 1994; González-Laxe, 2005). This principle is embedded in countless policy documents and legislation worldwide, including the Food and Agriculture Organisation (FAO)'s Code of Conduct for Responsible Fisheries (FAO, 1995), which provides a global reference framework to guide the sustainable use of fishery resources. The precautionary approach to fishery management often arises in the context of overfished stocks, as the prevention of overfishing is preferable to rebuilding fish stocks to a sustainable level, due to the economic and social costs that are incurred when fish stocks collapse (Caddy & Agnew, 2004). It is also relevant when decisions must be made concerning access granted to fishery users (FAO, 1996), a situation which often arises when marine protected areas (MPAs) are implemented close to a fishery (Lauck et al., 1998) or when granting access to users may cause environmental damage (FAO, 2003).

This thesis makes a contribution to our understanding of how these connections between marine habitats and fisheries influence the sustainable management of fishery resources. The objective of this thesis is to explore how information concerning the marine habitat may be incorporated into the decision-making processes of fishery management where fishing-induced habitat degradation is of concern, and the circumstances in which management tools usually used to achieve habitat improvements may generate fishery benefits. While there is a large literature concerning the interactions between habitat dynamics and fishery resource management (Armstrong & Falk-Petersen, 2008; Armstrong et al., 2017; Foley et al., 2012; Kahui et al., 2016; Reithe et al., 2014; Upton & Sutinen, 2005), little work has examined the role of habitat dynamics in controlling fishing mortality while rebuilding depleted fish stocks, nor explored how accounting for habitat degradation when implementing harvest allocations may impact the sustainability and overall performance of a fishery. Additionally, while there is an abundance of literature evaluating the impacts of conservation tools such as MPAs on nearby fisheries (Agardy et al., 2003; Dunne et al., 2014; Hilborn et al., 2004; Hughes et al., 2016; Smith et al., 2010), there is scarce empirical evaluation of the effect of a network of MPAs on the economic performance of nearby fisheries (Reimer & Haynie, 2018).

1.2. Controlling fishing mortality

Controlling the total level of harvest in a fishery is vital to managing fishery resources sustainably. Failure to do so leads to overfishing and eventual depletion of the marine resource (Gordon, 1954). This is because fishery resources, being rivalrous and non-excludable in consumption, are subject to the incentives of individual fishers, whereby each fisher attempts to obtain the greatest benefit from the resource, without regard for the negative impacts these attempts have on others or on the resource itself (Ostrom, 2008). The problems associated with these common good resources are well known in the fisheries literature (Gordon, 1954; Schaefer, 1957) and lead to the degradation of the resource and eventual stock collapse if left unchecked (i.e. the 'tragedy of the commons' (Hardin, 1968)). Methods to overcome these problems have included privatisation of the marine resource (Symes & Crean, 1995), including the granting of property rights such as individual transferable quotas (ITQs) (Costello et al., 2008), or the use of fishery management tools to control the level of harvest in a fishery the means of achieving this harvest, including fishing gears, fishery access, fishing seasons and spatial closures (FAO, 1995). Table 1 provides a non-exhaustive summary of fishery management methods sorted into two broad categories: input controls and output controls.[1]

[1] Adapted from Morison, 2004.

Table 1: Categories of fishery management methods to control fishing mortality

Input controls	Output controls
Limited entry (e.g. licencing)	Total allowable catch
Marine protected areas	
Time restriction (i.e. days at sea, seasonal restrictions)	Individual Transferable Quotas, catch shares
Gear restrictions	Territorial user rights fisheries TURFs

Input controls are those management tools which limit the amount of fishing effort or activity, and so indirectly impose control on the level of fishing mortality, while output controls impose direct limits on the amount of fish harvested. To explore the dynamics of habitat-fishery interactions and the way in which fishery management may be adapted in various contexts, this thesis focuses on fishery management methods from both categories, specifically total allowable catch (as assessed by harvest control rules), catch shares and marine protected areas.

1.2.1. Total allowable catch, harvest control rules and catch shares

The total allowable catch (TAC) of a fishery states the permitted level of harvest for a given species within a given time period. The TAC may be set according to a harvest control rule, which uses information regarding the state of the fish stock biomass to assess the appropriate TAC in a given time period (Agardy et al., 2011; Buxton et al., 2014; Punt, 2010). The current state of the fish stock biomass is assessed using a stock assessment model, which synthesises information concerning the life history, ecosystem features and current exploitation rates of fish species to estimate the level of biomass in a fishery (Cadrin & Dickey-Collas, 2015). Once this stock assessment process is complete, this information is applied to the harvest control rule and the TAC for that period is determined. Harvest control rules generally use two reference points when assessing TAC; the lower limit reference point and the target reference point. The lower limit reference point is the level of biomass below which a moratorium on fishing activity is declared. The target reference point is generally the level of biomass at which either the maximum sustainable yield (MSY) or the maximum economic yield (MEY) of the fishery may be achieved. If the fish stock biomass is found to be at at the target level, then the role of fishery management will be to set the TAC commensturate with this target. However, if the biomass is below the target level (that is, the stock is assessed as being overfished), the fish stock will require building to the target biomass. If a fish stock requires rebuilding, then a stock recovery plan may be implemented. A stock recovery plan normally involves defining a rebuilding target, selecting a trajectory for stock recovery and choosing the mechanisms through which the target is achieved along the selected trajectory (Caddy & Agnew, 2004). The mechanisms through which harvest is controlled will be highly contextual (Liu et al., 2016), and may be used in combination to minimise the risk that any one mechanism fails to adequately control fishing mortality (Hilborn et al., 2001; Stefansson & Rosenberg, 2005).

Once the TAC is determined and a stock recovery plan applied if relevant, a catch share program may also be applied to divide the harvest between individual fishers or fishing groups. Catch shares grant a level of access to fish stocks to either an individual user or group of users and thereby determine how potential benefit from the resources are shared. The mechanisms to manage catch shares in fisheries may include market-based mechanisms such as auctions of harvest rights or transferable individual quota systems, or be administrative in nature such as allocations determined by fishery management or set by legislation, or some combination of the two. Market-based catch shares such as individual transferable quotas (ITQs) are designed to grant exclusive and transferable rights over a portion of the TAC, which may then be bought and sold in an open market, which in theory results in an economic efficient allocation of harvest between fishery users (Arnason, 2008). In contrast, administratively-based allocation processes have the advantage of enabling managers to consider and maintain control over the wider interests of the community when determining allocations (Morgan, 1995), and are therefore a powerful tool in guaranteeing a certain level of harvest to vulnerable users (Bennett et al., 2018). Allocation processes based on administrative mechanisms have been used to guarantee access to fishery resources across sectors, including commercial, recreational and Indigenous, in order to maintain the social and cultural value of a fishery (Crowe et al., 2013; Islam & Berkes, 2016; Lynham, 2014; Sutinen & Johnston, 2003), and to promote the economic development of small coastal communities (Fina, 2011; Ginter, 1995; Haynie, 2014; Holland & Ginter, 2001).

1.2.2. Marine protected areas

Marine protected areas (MPAs) are a spatial management tool primarily designed to generate conservation and biodiversity improvements (Agardy et al., 2011; Wells et al., 2016), although increasingly MPAs are being adapted to also generate benefits for fisheries (Watson et al., 2014). MPAs control fishing mortality by controlling access to fish populations, with various restrictions on fishing activity imposed in different areas through MPA designations. MPAs are designated by the International Union for the Conservation of Nature (IUCN) according to their management objectives; categories range from category Ia, in which no extractive activity is permitted ('no-use MPAs'), to category VI, which permits sustainable exploitation of marine resources ('multi-use MPAs'). These IUCN categories control the level of access to marine resources within each MPA, and therefore dictate the extent to which fish stocks are available to harvest. Isolated no-use MPAs can be politically unfeasible (Smith et al., 2010) due to the loss of access to fishing grounds and the lengthy wait for fishery benefits to be generated in the form of spillover (Chollett et al., 2016). Partially in an effort to circumvent this problem, MPA networks comprised of collections of individual MPAs capable of operating synergistically, at various spatial scales and with a range of protection levels designed to achieve objectives that a single MPA cannot (IUCN-WCPA, 2008), are being established at increasing rates around the world. Rather than prohibiting access entirely, MPA networks enable both full protection for conservation and for fishers to have access fish stocks, and so have the potential to function collectively to facilitate both ecosystem and fishery improvements above those which might be expected from an individual MPA (Ballantine, 2014; Gaines et al., 2003; Gaines et al., 2010; Horigue et al., 2015; Roberts et al., 2001; Roberts et al., 2018).

1.3. Thesis aims and structure

The overarching aim of this thesis is to explore how the relationship between marine habitats and fisheries affects the sustainable management of fishery resources, and how these relationships may inhibit or improve fishery performance. These relationships are examined in three separate essays, each examining the links between habitat and fishery in three different contexts: first, where a depleted fish stock is targeted by a fishing gear which degrades the marine habitat, and requires rebuilding to achieve a sustainable level of biomass; second, where a fish stock is targeted by two competing user groups, one of which uses a highly damaging fishing gear; and third, where an adjacent fishery is subject to a network of MPAs implemented for conservation purposes.

The first essay (Chapter 2) focuses on the use of harvest control rules and no-use MPAs to control fishing mortality and so rebuild a depleted fish stock, in a fishery where fishing-induced habitat damage occurs. Using a bioeconomic model into which the two control mechanisms and habitat dynamics are incorporated, this essay focuses on alternative ways of exercising precaution in stock recovery plans to achieve stock rebuilding, while taking into consideration the economic and socio-economic objectives of fisheries management. In a parameterised simulation of this model, the efficacy of using these control mechanisms either separately or together to achieve stock rebuilding when habitat dynamics are present is examined, and the trade-offs between performance indicators when using these mechanisms is assessed.

The second essay (Chapter 3) assesses the optimal allocation of harvest across multiple user groups and how fishery outcomes are affected when catch shares are allocated suboptimally. We incorporate the harvest allocation process into a bioeconomic model in which fishing-induced habitat damage occurs and a single fish stock is targeted by two user groups characterized by fishing technologies of differing environmental impact. In a parameterised simulation of this model, we assess the consequences of omitting information concerning habitat degradation from the assessment of optimal harvest, how the habitat dynamics affect the optimal allocation of harvest between users, and the consequences to overall fishery performance when fishery management wishes to grant access to fish stocks to user groups who degrade the marine habitat through fishing activity.

The third essay (Chapter 4) presents an empirical analysis of the effect of an MPA network, established primarily for habitat conservation, on adjacent fisheries, using Australia's south-east marine reserve network as a case study. This essay uses difference-in-difference modelling to isolate the effect this network had on the performance of nearby fisheries, using a panel of data comprised of 8 fisheries and 39 species with a time series spanning the years 2001-2015. Using this methodology, this essay examines the effect of this network on catch and the gross value of production for adjacent fisheries, while controlling for major fishery management changes which occurred during the same time period. This essay also explores the effect this network may have had for commercially valued species in particular.

The final chapter (Chapter 5) concludes this thesis and suggests avenues for future research.

Chapter 2: The role of precaution in stock recovery plans in a fishery with habitat effect

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Note: This essay uses the term 'marine reserve' in place of 'no-use MPA'

2.1. Introduction

The precautionary principle of natural resource management was first articulated in the United Nations' World Charter for Nature in 1982 and has since been integrated into many legally binding international treaties, as well as the legislation of numerous countries. In broad terms, the precautionary approach `expresses a desire to prevent damage to the environment before it occurs and, if damage has previously occurred, not to postpone or avoid taking action due to scientific uncertainty. The precautionary principle in the context of fisheries management was first introduced in the FAO Code of Conduct for Responsible Fisheries (1995). In this context, the objective of the precautionary principle is to prevent resource and environmental degradation and to rebuild depleted fish stocks, while taking into consideration the economic and socio-economic requirements of fisheries (González-Laxe, 2005). The prevention of fish stock collapse, and so the avoidance of the economic and social costs that accompany stock collapse, therefore provide a strong incentive to apply the precautionary principle to fisheries management (Caddy & Agnew, 2004). The current depleted state of a large proportion of fisheries (FAO, 2016) means that understanding the outcome of applying the precautionary principle to fisheries rebuilding is highly relevant for many countries.

Stock recovery plans have been applied to fish stocks around the world with varying levels of success (Murawski, 2010). A stock recovery plan involves defining a rebuilding target, selecting a trajectory for stock recovery and choosing the mechanisms through which the target is achieved along the selected trajectory (Caddy & Agnew, 2004). The identification of the right mechanism to control fishing mortality is, therefore, central to the success of the rebuilding plan. Two mechanisms commonly used to control fishing mortality are harvest control rules and no-take marine reserves. Harvest control rules manage fishing mortality by directly setting limits on the amount of biomass that can be harvested in any given time period of the rebuilding trajectory (Agardy et al., 2011; Buxton et al., 2014; Punt, 2010). No-take marine reserves, on the other hand, control fishing mortality by protecting some proportion of the biomass from harvest (Lester et al., 2009). The challenges of designing and implementing harvest control rules, and the way in which the precautionary principle is integrated into the development of harvest control rules has been widely discussed in the literature (Cadrin & Pastoors, 2008; Hilborn et al., 2001; Kvamsdal et al., 2016; Punt, 2006). Moreover, no-take reserves have been shown to be a means of applying the precautionary approach in fisheries management, while mitigating factors such as scientific uncertainty, management error and habitat damage (Lauck et al., 1998; Mangel, 2000; Roberts et al., 2005).

The limitations of using the two control mechanisms individually to control fishing mortality in recovering a fishery are widely acknowledged in the literature. For example, implementing the precautionary approach through harvest control rules alone may fail to limit fishing mortality at a desired level, due to the lack of wider ecosystem considerations inherent in such rules (Cadrin & Pastoors, 2008). Further, management's limited capacity to frequently revise the rules may lead fishery managers to adopt a more precautious harvest control rule at the beginning of the rebuilding plan, which will heighten the trade-offs between competing fishery objectives, such as the maintenance of short-term harvest and the stock rebuilding period

(Wetzel & Punt, 2016). Despite these limitations, there are few studies that look at implementation of the precautionary principle in stock recovery plans using the two mechanisms together. Exceptions are Little et al., 2011 and Yamazaki et al., 2015 who have studied the complementarity of harvest control rules and no-take marine reserves. In particular, the latter shows that the use of harvest control rules and no-take reserves together allows a fishery manager to design a rebuilding plan which can hasten the speed of stock recovery without reducing the profitability and annual harvest of the fishery.

The aim of this paper is to extend this literature by exploring how the precautionary principle should be implemented through the two control mechanisms both individually and in concert in a stock recovery plan where the fishery is subject to fishing-induced habitat degradation. Previous studies assume no relationship between fishing activity and habitat damage, thereby assuring the rebuilding of fish stocks in response to reduced fishing pressure. However, the collapse of a fishery is often characterised by both the depletion of fish stocks and the degradation of habitat, the latter of which may prevent stock rebuilding due to slow regeneration processes or irreversible habitat loss. The connection between fishing activity and habitat damage is the subject of increasing study and concern (Hiddink et al., 2011; Kahui et al., 2016; Shephard et al., 2010; Turner et al., 1999). Moreover, previous literature has examined habitat and marine reserve interactions under different regulatory regimes (Akpalu & Bitew, 2014; Akpalu & Bitew, 2011; Moeller & Neubert, 2012, 2015; Reithe et al., 2014; Upton & Sutinen, 2005). To the best of our knowledge, however, there has been no study to date of how marine reserves and harvest control rules may be used together in stock recovery plans where fishing-induced habitat damage is a feature of the fishery.

To this end, we develop a bioeconomic model of a fishery where the carrying capacity of the population biomass is impacted by fishing (i.e., habitat effect) and the regeneration of the habitat does not occur immediately but requires time. We consider alternative stock rebuilding strategies characterised by different levels of precaution exercised jointly or separately through the harvest control rule and marine reserve. The performance of alternative stock rebuilding strategies is assessed against three indicators which broadly correspond to the biological, economic and socio-economic objectives of fisheries management. Using the three performance indicators we identify and assess trade-offs between potentially conflicting fisheries objectives where different levels of precaution are exercised through harvest control rules and marine reserves. We further explore the possibility of maximising the economic and socio-economic indicators while meeting the constraint of a mandated time limit for stock rebuilding. Stock recovery plans generally aim to rebuild fish stocks within a prescribed time period (see, for example, the Magnuson-Stevens Fisheries Conservation and Management Act of the USA). However, the needs of fishers and other stakeholders to earn income and maintain employment in both the short- and long-term are also important considerations (Hilborn et al., 2001; Mardle & Pascoe, 2002). An understanding of how stock rebuilding plans may be designed to minimise the trade-offs between these competing objectives is, therefore, of high importance.

2.2. Methods

2.2.1. Biomass Dynamics

Specification of the biomass dynamics is based on previous studies (Conrad, 1999; Grafton et al., 2006; Hannesson, 1998; Sanchirico & Wilen, 2001; Yamazaki, Jennings, et al., 2015). The total population, x_t, consists of two subpopulations such that $x_t = x_t^H + x_t^R$, where x_t^H is the harvest population and x_t^R is the reserve population, and the subscription t = 0,1,2,… denotes the time index. The size of the reserve is defined as the proportion of the population carrying capacity that is not exposed to fishing and is determined by the parameter $s \in [0,1]$.

The biomass dynamics of the two subpopulations are specified as:

$$x_{t+1}^H = x_t^H + G^H(x_t^H, K_t, s) + T(x_t^H, x_t^R, K_t, s) - h_t \quad (1)$$

$$x_{t+1}^R = x_t^R + G^R(x_t^R, K_t, s) - T(x_t^H, x_t^R, K_t, s) \quad (2)$$

where h_t is the total harvest at time t where harvest of the reserve population is prohibited. Each subpopulation has its own specific growth function, $G^H(\cdot)$ and $G^R(\cdot)$, and the annual growth of each population depends on the population biomass and carrying capacity[2] at time t, x_t^j and K_t, as well as the reserve size, s. The two subpopulations are linked by the transfer function $T(\cdot)$. The growth functions for the harvest and reserve populations are specified as:

$$G^H(x_t^H, K_t, s) = rx_t^H\left(1 - \frac{x_t^H}{(1-s)K_t}\right) \quad (3)$$

$$G^R(x_t^R, K_t, s) = rx_t^R\left(1 - \frac{x_t^R}{sK_t}\right) \quad (4)$$

where r is the intrinsic growth rate.[3] The transfer function takes the following form:

$$T(x_t^H, x_t^R, s) = m(1-s)\left(\frac{x_t^R}{s} - \frac{x_t^H}{(1-s)}\right) \quad (5)$$

where m is the transfer coefficient which measures the strength of the links between reserve and harvest subpopulations. We base our assumptions on population transfer on empirical evidence which suggests that fish migration is likely to be density-dependent and a function of reserve size (Abesamis & Russ, 2005; Goñi et al., 2010; Kramer & Chapman, 1999). The pre-multiplicative term, (1-s), ensures that the spillover between reserve and harvest populations

[2] As we assume two subpopulations of a single stock, the two subpopulations share the same population carrying capacity K_t.
[3] In the extreme case where s = 0 or s = 1, the carrying capacity of one subpopulation will become zero, so that the entire biomass will be either exposed or not exposed to harvesting.

becomes smaller with increased reserve size (Grafton et al., 2006; Hannesson, 1998; Kramer & Chapman, 1999).

2.2.2. Habitat Effect and Dynamic Carrying Capacity

Ecosystem externalities occur when the act of harvesting fish impacts the underlying processes that govern the ecological system (Ryan et al., 2014). Ecosystem externalities may include adverse impacts on the productivity of stocks through damage to the habitat (Janmaat, 2011). This paper incorporates the effect of fishing-induced habitat changes on fish biomass through the population carrying capacity. That is, the population carrying capacity increases when the habitat recovers and decreases when the habitat is damaged due to fishing. Following Upton & Sutinen (2005) and Udumyan et al., (2010), the dynamics of the population carrying capacity is specified as follows:

$$K_{t+1} = K_t + H(K_t) - D(E_t, K_t, s) \tag{6}$$

where $H(\cdot)$ is the logistic growth rate of the carrying capacity, which increases as the habitat of the fish stock increases, and $D(\cdot)$ is the reduction in carrying capacity due to the habitat damage caused by fishing. The habitat growth and damage functions are specified as:

$$H(K_t) = \rho K_t \left(1 - \frac{K_t}{K_{MAX}}\right) \tag{7}$$

$$D(E_t, K_t, s) = \gamma E_t (1-s) K_t \tag{8}$$

where ρ is the intrinsic regeneration rate, K_{MAX} is the population's maximum possible carrying capacity, E is the fishing effort and γ is the rate of degradation caused by fishing effort. The damage function indicates that damage is negatively associated with the reserve size, since harvest of the reserve population is prohibited and, therefore, the damage caused to the habitat by fishing decreases as reserve size increases, *ceteris paribus*. Given the dynamics of the carrying capacity of the habitat in (6)-(8), the cases of no habitat effect with constant carrying capacity, and habitat effects which preclude regeneration of habitat, can be considered as special cases where ρ and γ are equal to zero.

2.2.3. Harvest and Profit Functions

We use a modified version of the Schaefer harvest function given as:

$$h_t = \frac{q}{(1-s)} E_t x_t^H \tag{9}$$

where q is the catchability coefficient. The specification of the harvest function implies that, holding the value of x_t^H constant, the rate of fishing mortality per unit of fishing effort (CPUE) increases with the reserve size, i.e., $\partial[h_t/E_t]/\partial s > 0$. In other words, given the size of harvest population, increasing the reserve size increases the population density of the harvest population, resulting in a higher level of CPUE. This model assumption is validated empirically in a variety of environments (McClanahan & Kaunda-Arara, 1996; Russ et al., 2004).

Net profit at time t is given as:

$$\pi_t = p(h_t)h_t - cE_t \tag{10}$$

where $p(h_t)$ is the inverse demand function and c is the cost per unit of fishing effort. The inverse demand is given as:

$$p(h_t) = ah_t^{\varepsilon} \tag{11}$$

where a is a parameter and $1/\varepsilon$ is the price elasticity of demand. Rearranging equations (9)-(11) yields the net profit function in terms of the harvest, harvest population and reserve size, such that:

$$\pi(h_t, x_t^H, s) = ah_t^{\varepsilon+1} - \frac{bh_t(1-s)}{x_t^H} \tag{12}$$

where $b = c/q$. This profit function is conformable to the one used elsewhere (Clark, 1990; Grafton et al., 2009). Given the net profit function, the net present value of the fishery is given as:

$$NPV = \sum_{t=0}^{\infty} \frac{1}{(1+\delta)^t} \pi(h_t, x_t^H, s) \tag{13}$$

where $\delta \in [0,1]$ is the discount rate.

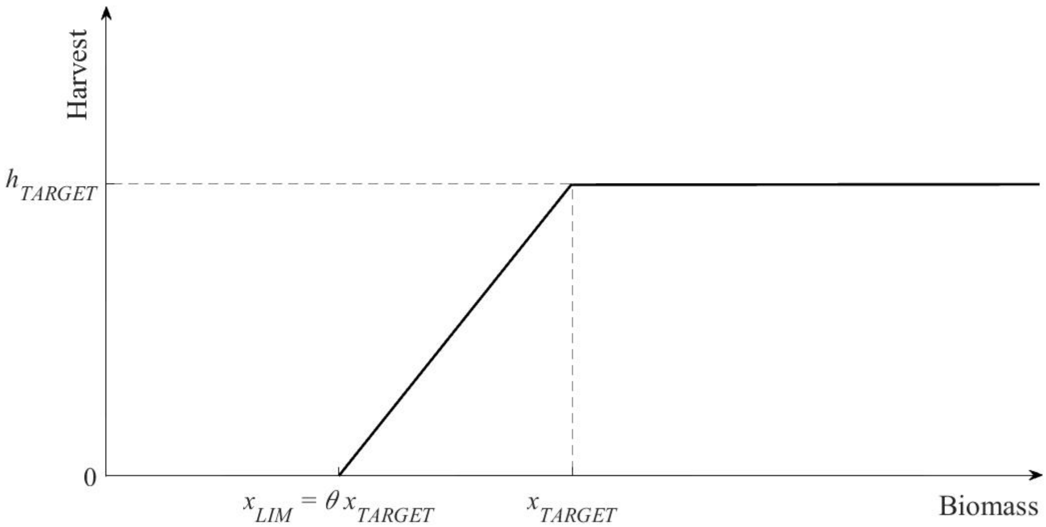
2.2.4. Harvest Control Rule

To explore alternative stock rebuilding strategies characterised by different levels of precaution, exercised jointly through the harvest control rule and marine reserve size (s), we consider a generic harvest control rule. The generic rule is informed by the annual assessment of total population biomass, x_t, to set a harvest limit in each year, h_t, such that:

$$h_t = \begin{cases} 0 & \text{if} \quad x_t \le x_{LIM} \\ \dfrac{h_{TARGET}}{x_{TARGET} - \theta x_{TARGET}} x_t^H - \dfrac{\theta h_{TARGET}}{(1-\theta)} & \text{if} \quad x_{LIM} < x_t < x_{TARGET} \\ h_{TARGET} & \text{if} \quad x_t \ge x_{TARGET} \end{cases} \tag{14}$$

The target level of total population biomass is denoted as x_{TARGET}, and h_{TARGET} is the level of harvest associated with this target biomass (Figure 1). We assume that there is a minimum level of biomass that is deemed acceptable for the fishery to be open and this lower limit of the biomass is denoted by x_{LIM}. The harvest control rule in (14) implies that, for any observed level of total biomass below x_{LIM}, no harvest is permitted, $h_t = 0$. For any observed level of biomass greater than x_{TARGET} the annual catch limit is set at h_{TARGET}. However, for any level of biomass between x_{LIM} and x_{TARGET}, the annual harvest depends on the parameter θ, which is referred to as *precautionary parameter* in this paper and $\theta \in [0.1,1]$. We set the lowest value θ can take at 0.1 to reflect the fact that there must be a certain amount of biomass left in order for the stock to rebuild. The lower limit of biomass is assumed to be proportional to the target biomass; that is, $x_{LIM} = \theta x_{TARGET}$, and given this, varying degrees of precaution exercised through the harvest control rule can be modelled by applying different values of the precautionary parameter. In other words, the parameter θ is referred to as 'precautionary' as it controls how low the biomass is permitted to fall before the fishery closes, and so is an indication of the precaution fishery management exercises against stock collapse through the harvest control rule.

Figure 1. Harvest control rule. The lower limit biomass, x_{LIM} , is proportional to the biomass target, x_{TARGET}. The precautionary parameter θ determines the lower limit biomass is permitted to reach before harvest stops, such that $x_{LIM} = \theta x_{TARGET}$.



The harvest control rule given in (14) is compatible with some existing harvest strategies, such as those set out in the Commonwealth Harvest Strategy Guidelines in Australia and the Magnuson-Stevens Fishery Conservation and Management Act in the U.S. The lower biomass limit, x_{LIM}, is set at the policy level, and specifies the level below which the risk of extinction to the stock is considered too great, with the rebuilding trajectory being incidental to this lower limit.[4] Fishery managers use these guidelines to inform their fishery management strategies, with consideration to the needs of particular target species and stakeholders.

2.2.5. Stock Rebuilding Strategies and Performance Indicators

A combination of the precautionary parameter and reserve size, (θ, s), defines a stock rebuilding strategy with a given level of precaution. Precaution in the harvest control rule is controlled through the precautionary parameter, θ; as this value increases, the control rule becomes more precautious. Similarly, the reserve size, s, specifies the proportion of the carrying capacity set aside as a no-take reserve; as this value increases, precaution in fishery management increases (Lauck et al., 1998).

We present our results against three performance indicators which broadly correspond to the biological, economic and socio-economic objectives of fishery management, namely the number of years taken to rebuild biomass to the target level (i.e., the stock rebuilding period); the NPV of the fishery, and the average harvest over the first ten years of the stock rebuilding period. Fish stocks are considered to be rebuilt when biomass first reaches 95% of the biomass target. For the economic indicator, the NPV of the fishery given in (14) is calculated over 100 time periods. We include the average 10-year harvest as the short-term socio-economic indicator to account for the fact that a rebuilding strategy that achieves stock rebuilding within a given timeframe or that maximises NPV of the fishery over the long run may not be viable because of short to medium-term reductions in the harvest.[5]

[4] In Australia and the U.S., the lower limit is set to 50% of x_{MSY}. This corresponds to a θ value of 0.5 in this model.

[5] We use the average harvest as a socio-economic performance indicator because maintaining harvests in the short term is positively related to employment and other means of support (Hilborn, 2007a). Other performance indicators may have included calculating the additional profit generated by the marine reserve, the social value of improving marine habitats (Pelletier et al., 2005). Assessing the stock rebuilding strategies against a wider range of performance indicators is a potential extension of this paper.

2.2.6. Parameters

The benchmark parameter values used in the simulations are presented in Table 1. The bioeconomic model is simulated with the estimated parameter values reported in Grafton et al., 2009 for the northern cod fishery of Newfoundland and Labrador. This fishery was once one of the world's largest capture fisheries, before stocks collapsed in the early 1990s. The collapse of these cod stocks was due to several factors, including overfishing and mismanagement, although the effects of habitat degradation are thought to partially explain why the cod stocks have failed to recover so far (McCain et al., 2016). The purpose of this analysis is not to conduct a case study, or provide an empirical evaluation of this specific fishery, but to develop insights into the way in which the application of the precautionary principle changes in the presence of fishing-induced habitat damage, where harvest controls and marine reserves are the management instruments of choice. The sensitivity of our results to changes in key parameter values is explored through scenario analysis.

Table 1. Benchmark parameters

Parameter	Description	Value
δ	Time discount rate (per year)†	0.05
a	Price parameter†	0.35
ε	Reciprocal of the price elasticity of demand †	-0.3
b	Cost parameter†	0.20
r	Intrinsic growth rate (per year)†	0.27067
q	Catchability coefficient‡	0.0825
K_{MAX}	Maximum carrying capacity (000 tonnes)†	3200
K_0	Initial carrying capacity (000 tonnes)††	2050
m	Transfer coefficient††	0.100
ρ	Growth rate of habitat (per year)††	0.100
γ	Damage rate of habitat (per year)††	0.010
x_0	Initial biomass (000 tonnes)††	526
x_{MSY}	Maximum Sustainable Yield Biomass (000 tonnes) ††	1600

† The parameter values are taken from Grafton et al., 2009.
†† The parameter values are calculated or set by authors.
‡ The catchability coefficients of the five cod fishery areas for the years 1970 to 1980 are averaged based on the estimates provided by Pinhorn, 1988.

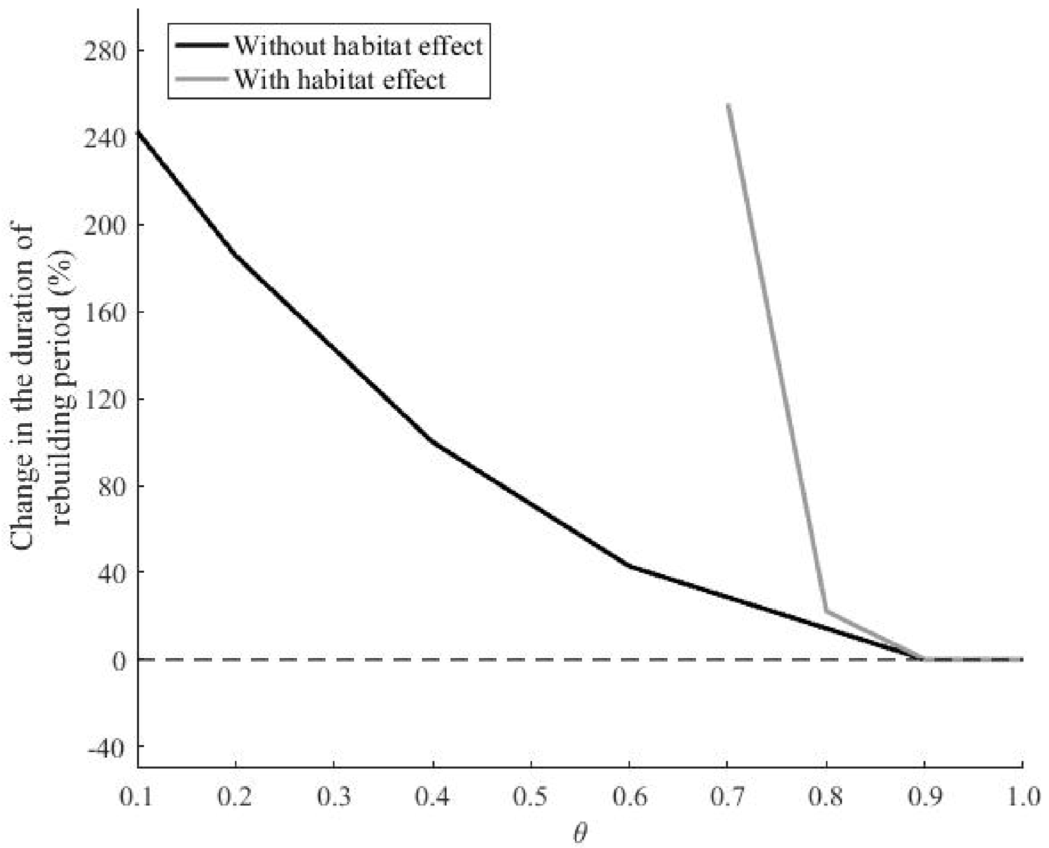
As the focus of this paper is stock rebuilding, we set the initial biomass as 526 thousand tonnes, which was the estimated biomass in 1977 when Canada assumed jurisdiction over the entire fishery (Grafton et al., 2009). We set the biomass target at x_{MSY}, the biomass which supports the maximum sustainable yield. The maximum sustainable yield is calculated by maximising the surplus production for the fishery, and we use this to locate the corresponding x_{MSY}. Habitat effects are not included in the calculation of the maximum sustainable yield, implying that fishery managers do not explicitly account for the habitat effect in the determination of either the biomass target or in the formulation of the harvest control rule in (14). For the initial condition of the habitat, we assume that the habitat has degraded as a result of previous over-exploitation and set the initial carrying capacity at 2.05 million tonnes or 64% of the maximum carrying capacity. Where we assume that no habitat effect exists, the carrying capacity is constant at the maximum level of 3.22 million tonnes as there is no habitat-fishery linkage. The sensitivity of the results to the initial value of the carrying capacity is carried out in scenario analysis.

2.3. Results

2.3.1. Precaution through Harvest Control Rule and Stock Rebuilding Period

Figure 2 shows the stock rebuilding period for different values of the precautionary parameter, θ, relative to when the most precautious harvest strategy (θ = 1) is adopted and no marine reserve is established (s = 0). The figure shows that, where it is assumed there are no habitat effects, it is possible to rebuild the fishery regardless of the level of precaution exercised through the harvest control rule, but the stock rebuilding period decreases with an increased level of precaution. For example, a precautionary parameter value of θ = 0.1 has a rebuilding time 240% longer than the baseline, while a more precautionary value of θ = 0.8 has a rebuilding time 15% longer. In other words, where no habitat effects exist, the stock rebuilding period is longest when the harvest control rule permits the highest possible level of harvest and decreases as the permitted level of harvest decreases. However, if our assumption of constant habitat is incorrect, and in fact habitat is degraded by fishing, the fish stock fails to rebuild to its biomass target level for all but the most precautious harvest strategy. More particularly, when habitat effects are present, the fishery will not achieve rebuilding unless the value of the precautionary parameter is greater than 0.7. This represents the minimum level of precaution that needs to be exercised through the harvest control rule for successful rebuilding when there exist habitat effects and no reserve is used in the rebuilding strategy.

Figure 2. Relative effect of harvest control rule precaution on the stock rebuilding period without habitat effect (black line) and with habitat effect (grey line). The baseline case is the most precautious harvest strategy (θ = 1) and no reserve (s = 0).

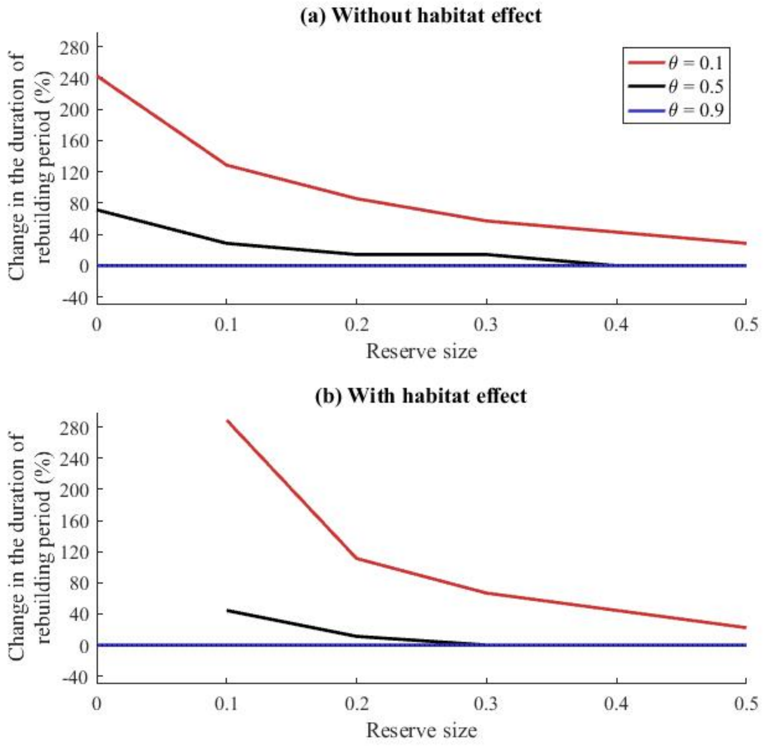


Moreover, in cases where the fishery does achieve rebuilding in the presence of habitat effects, the stock rebuilding period is much longer compared to the stock rebuilding period calculated in the absence of habitat effects. For example, where the precautionary parameter equals 0.7, the stock rebuilding period is 240% higher than the baseline, compared to 20% higher in the absence of a habitat effect. These results suggest that the habitat effect reduces the ability of the harvest control rule to rebuild the depleted fishery, but also that the stock rebuilding period is highly sensitive to changes in habitat even when the harvest strategy is highly precautious. However, as precaution increases, the difference in the stock rebuilding period due to the habitat effects becomes negligible. For example, if the precautionary parameter is greater than 0.85, the stock rebuilding period is identical in both cases.

2.3.2. Substitutability of Marine Reserves and Harvest Strategies for Stock Rebuilding

We have seen that the habitat effect can cause failure of a depleted stock to rebuild or lengthening of the stock recovery time, unless a maximum level of precaution is applied in the harvest control rule. We now explore whether, and to what extent, precaution introduced in the harvest control rule can be substituted for precaution exercised through the use of marine reserves for stock rebuilding.

Figure 3. Relative effect of reserve size on the stock rebuilding period. The baseline case is the most precautious harvest strategy (θ = 1) and no reserve (s = 0). Panel (a): without habitat effect and Panel (b): with habitat effect. Three levels of precaution in the harvest control rule are applied: θ = 0.1 (red); θ = 0.5 (black); and θ = 0.9 (blue).



Where fishing does not have an adverse impact on habitat, the establishment of a marine reserve shortens the duration of the stock rebuilding period, particularly for the least precautious harvest strategy (Figure 3a). As noted above, when there is no reserve, the stock rebuilding period is 240% longer than the baseline for a precautionary parameter of 0.1. Once a reserve size of 10% is established, however, this rebuilding period halves to 120%, and continues to decrease as the reserve size increases. Nevertheless, where a relatively high level of precaution is already exercised through the harvest control rule (e.g., θ = 0.9), a marine reserve offers no additional benefits to the stock rebuilding period. For example, where the precautionary parameter is 0.9, the rebuilding period remains the same as the baseline as the reserve size increases. These results suggest there exists substitutability between the level of precaution applied through the harvest control rule and marine reserve in a rebuilding strategy. In effect, there is a range of combinations of the precautionary parameter and reserve size which result in the same stock rebuilding period.

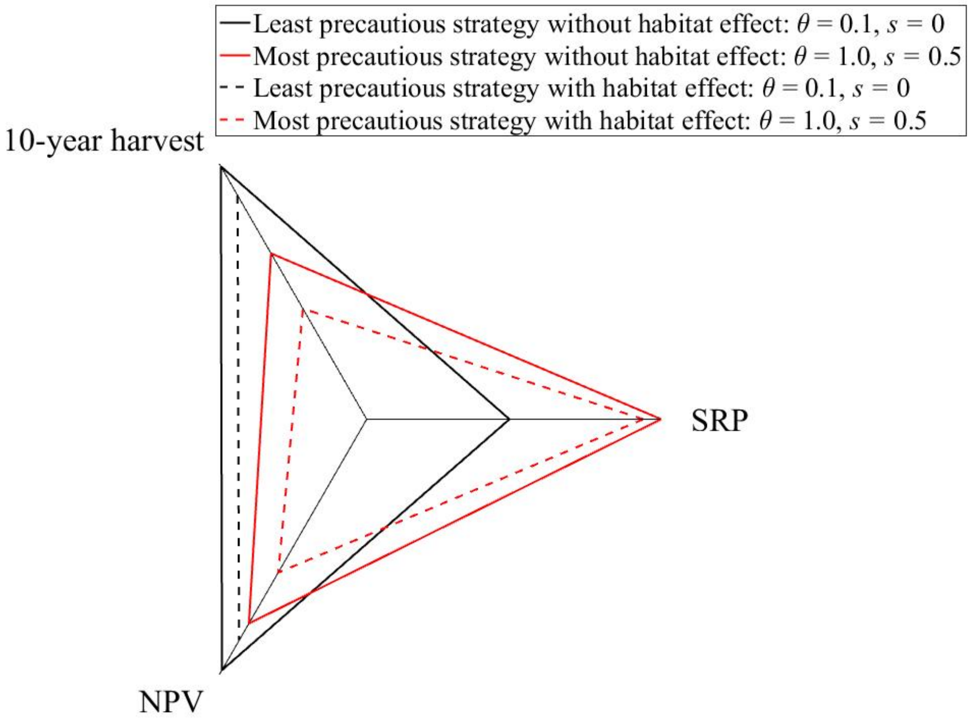
Where the habitat effect exists, we observe the same negative relationship between the stock rebuilding period and reserve size, particularly when precaution through the harvest control rule is relatively low (Figure 3b). Moreover, as shown in our earlier results, the existence of habitat effects may prevent the rebuilding of the fishery unless the most precautious harvest strategy is used, but Figure 3b shows that increasing the reserve size can offset this and enable stock rebuilding to be achieved for all precautionary parameters applied to the harvest control rule. For example, the precautionary parameter values of 0.1 and 0.5 fail to achieve rebuilding when there is no reserve established, but rebuilding is achieved regardless of the value of the precautionary parameter once a reserve of 10% or greater is established. In these cases, where a marine reserve is used as part of the stock rebuilding strategy, there no longer exists a minimum level of precaution exercised through the harvest control rule to ensure successful rebuilding. However, increasing the reserve size offers a decreasing marginal benefit to the stock rebuilding period. That is, as the reserve size increases further, the reduction in the recovery time becomes smaller, and eventually goes to zero, and this result holds independent of the habitat effect. For example, where the precautionary parameter is 0.5, reserve sizes greater than 30 or 40% have no additional effect on the stock rebuilding period.

2.3.3. Conflicting Management Objectives and Trade-offs between Performance Indicators

We examine the trade-offs between conflicting management objectives[6] by first examining the way in which the nature and magnitude of the trade-offs between performance indicators are affected when switching between the least precautious (θ = 0.1, s = 0) and most precautious rebuilding strategy (θ = 1.0, s = 0.5). Figure 4 shows how the two rebuilding strategies with strongly contrasting levels of precaution perform against the three performance indicators: stock rebuilding period (SRP), average 10-year harvest, and NPV of the fishery. These trade-offs are presented in the form of a spider plot, in which each axis corresponds to the outcome for each indicator. Average 10-year harvest and NPV of the fishery increase when moving from right to left along their axes, while the SRP decreases as with movement away from the origin. The value of each performance indicator is standardised such that the figure presents the relative outcomes between the least and most precautious strategies as well as across the two cases of habitat effects.

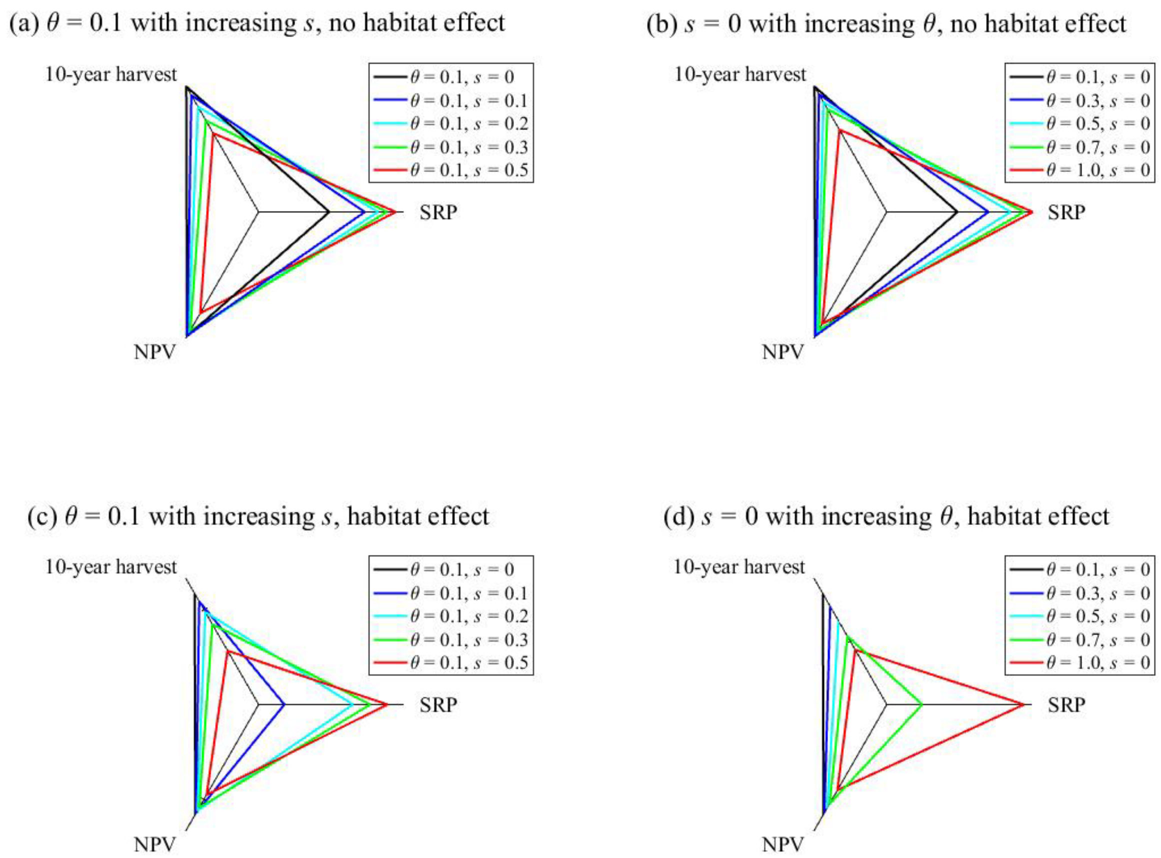
[6] We note that the three performance indicators considered in this paper are correlated to each other. For example, a longer stock rebuilding period results in the fishery benefits being achieved at a later year so that the NPV of the fishery is likely to be lower. One common approach in economics is to find the harvest strategy that maximises the NPV of the fishery and this optimal strategy informs "appropriate" trade-offs between different objectives. In this paper, we instead pre-determine performance indicators, and alternative rebuilding strategies are then evaluated against these indicators. In this approach, opposing effects on different performance indicators are interpreted as a trade-off as each indicator corresponds to different management objectives. In other words, the trade-off here means that a choice needs to be made by management across these biological, economic and socio-economic indicators.

Figure 4. Trade-offs in rebuilding strategies without habitat effect (solid line) and with habitat effect (dotted line). Black lines represent the least precautious harvest strategy with no reserve (θ = 0.1 and s = 0). Red lines represent the most precautious harvest strategy with the 50% reserve (θ = 1.0 and s = 0.5).



A shift from the least precautious to most precautious strategy shows a clear trade-off between the SRP and NPV of the fishery regardless of the presence of habitat effects. The choice of whether to implement the least or most precautious strategy also creates a trade-off between the SRP and 10-year harvest. That is, increasing the level of precaution in the stock rebuilding strategy through both the harvest control rule and marine reserves results in a faster rebuilding time, but this is achieved at the expense of declines in 10-year harvest and NPV of the fishery. This negative relationship between the performance indicators occurs regardless of the existence of habitat effects, but habitat effects amplify the magnitude of the trade-off. This is evident from the fact that, in the fishery with habitat effects, the switch from the least to the most precautious strategy results in a substantial improvement in the SRP (with the fishery that previously failed to rebuild now achieving rebuilding) with relatively modest reductions in both 10-year harvest and NPV. Moreover, for both the most and least precautious strategies, the presence of habitat effect causes a decline in the biological, economic and socio-economic outcomes of the fishery, but the relative effects on these performance indicators are greater when the least precautious rebuilding strategy is implemented.

Figure 5. Trade-offs in rebuilding strategies. Panels (a) and (b): without habitat effect. Panels (c) and (d): with habitat effect. Different levels of precaution are implemented through (a,c) an increase in the reserve size, *s*, or (b,d) an increase in the precautionary parameter in the harvest control rule, *θ*.



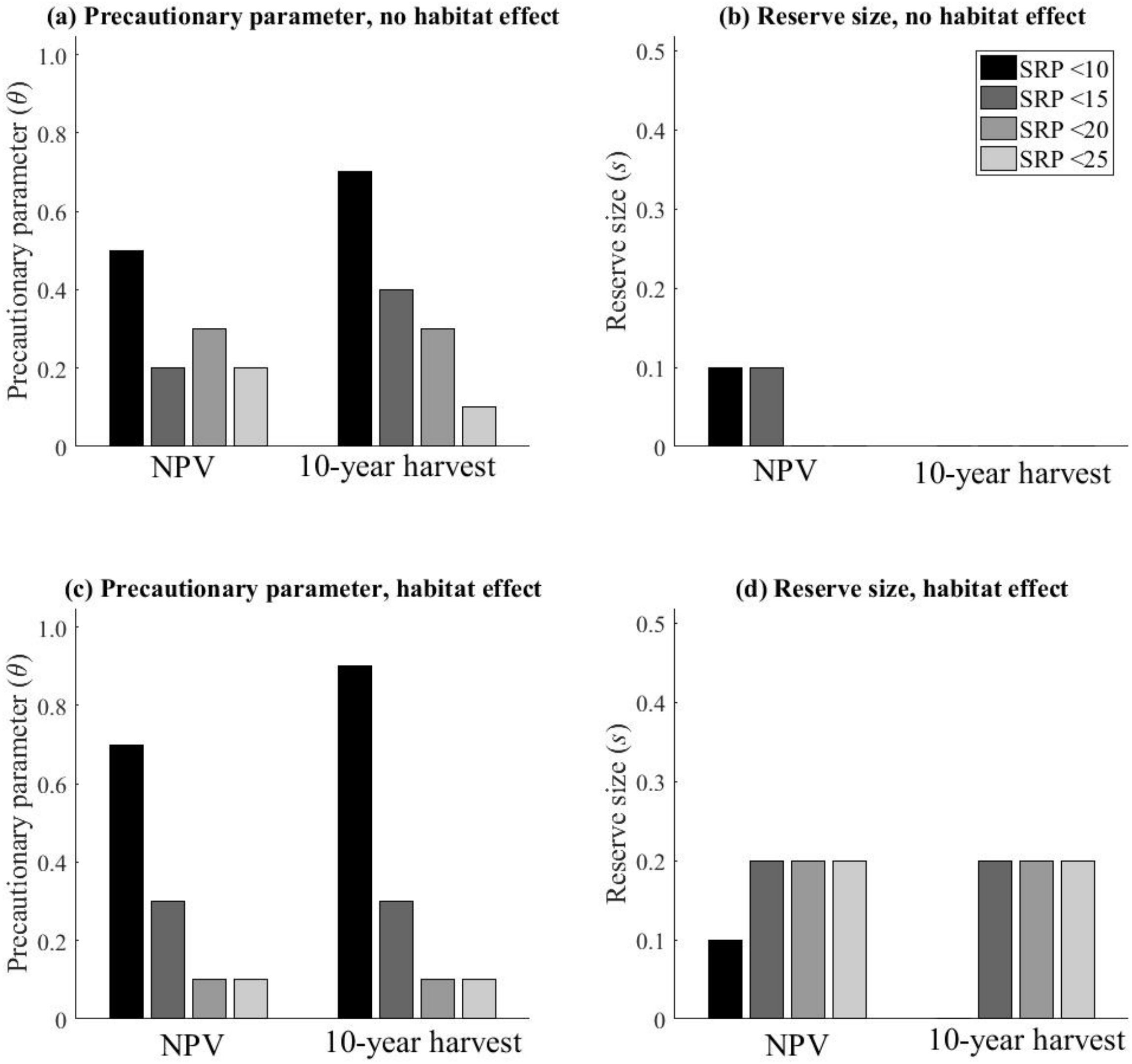
Instead of implementing either the most or least precautious rebuilding strategy through both the harvest control rule and marine reserves, different levels of precaution can be exercised solely through one of the instruments. Figure 5 shows how the nature of trade-offs between the performance indicators is affected by habitat effects when precaution is exercised through either the harvest control rule (θ) or marine reserve (s). The value of each performance indicator is standardised so that it allows a comparison across different panels. Where habitat effects are absent, as in the previous case when both instruments were used, there exist trade-offs between the performance indicators as precaution increases, regardless of which instrument is used. That is, an increase in precaution exercised through one of the instruments hastens the speed of stock rebuilding but this is achieved at the expense of declines in the 10-year harvest and NPV of the fishery. The trade-off is particularly observed between the SRP and 10-year harvest, while NPV of the fishery is only moderately affected by increasing the level of precaution through either the harvest control rule or marine reserve.

We observe similar trade-offs where habitat effects exist, however, habitat effects amplify the magnitude of the trade-offs and the relative impacts of habitat effects differ depending on which instrument is used to apply precaution in the stock rebuilding strategy. In particular, the trade-off between the SRP and 10-year harvest is the most prominent when precaution is exercised solely through the harvest control rule and there is no reserve established. In other words, where habitat effects are present, increasing the precautionary parameter from 0.1 to 0.7 or greater enables a faster and successful rebuilding without a reserve, yet the cost is a decline in the 10-year harvest by 38 to 50%.

2.3.4. Rebuilding Mandate and Optimal Rebuilding Strategy

Given the trade-offs between the performance indicators, and the way in which these trade-offs are affected by the manner in which precaution is implemented through the two instruments, we can determine a rebuilding strategy that simultaneously maximises the NPV of the fishery and average 10-year harvest while meeting the constraint of a mandated time limit for stock rebuilding. We refer to such a combination of the precautionary parameter and the reserve size, (θ, s), as the optimal rebuilding strategy. Figure 6 shows optimal strategies for four stock rebuilding constraints, namely 10, 15, 20 and 25 years.

Figure 6. Optimal rebuilding strategies (θ, s) that maximize the NPV of the fishery and 10-year average harvest for a given rebuilding mandate. Panels (a) and (b): without habitat effect. Panels (c) and (d): with habitat effect.



Where there are no habitat effects, there is one strategy which simultaneously maximises both the NPV of the fishery and 10-year harvest: the combination (0.3, 0) for the rebuilding constraint of less than 20 years. If the rebuilding mandate is either greater or less than 20 years, there is no unique combination of the precautionary parameter and reserve size that maximises both the NPV of the fishery and the 10-year harvest, but a single objective is maximized. For example, the rebuilding strategy which rebuilds the fishery in less than 10 years and maximises NPV is (0.5, 0.1), while the strategy which maximises the 10-year harvest for the same rebuilding constraint is (0.7, 0). This suggests that, in meeting a given rebuilding mandate, it is not always possible to maximise both the NPV of the fishery and 10-year harvest simultaneously, but there is a trade-off between the two objectives in choosing a rebuilding strategy. The possibility that this trade-off will exist, and of there being no jointly optimal strategy, is unaffected when habitat effects are present, but there is a greater number of strategies which simultaneously maximise both the NPV and 10-year harvest for the fishery and meet the rebuilding mandate. That is, if the rebuilding constraint is 15 years or longer, precautionary strategies which maximise the NPV of the fishery also maximise the 10-year harvest. Therefore, where habitat effects exist and a relatively relaxed rebuilding constraint is mandated, it is possible to achieve both objectives simultaneously.

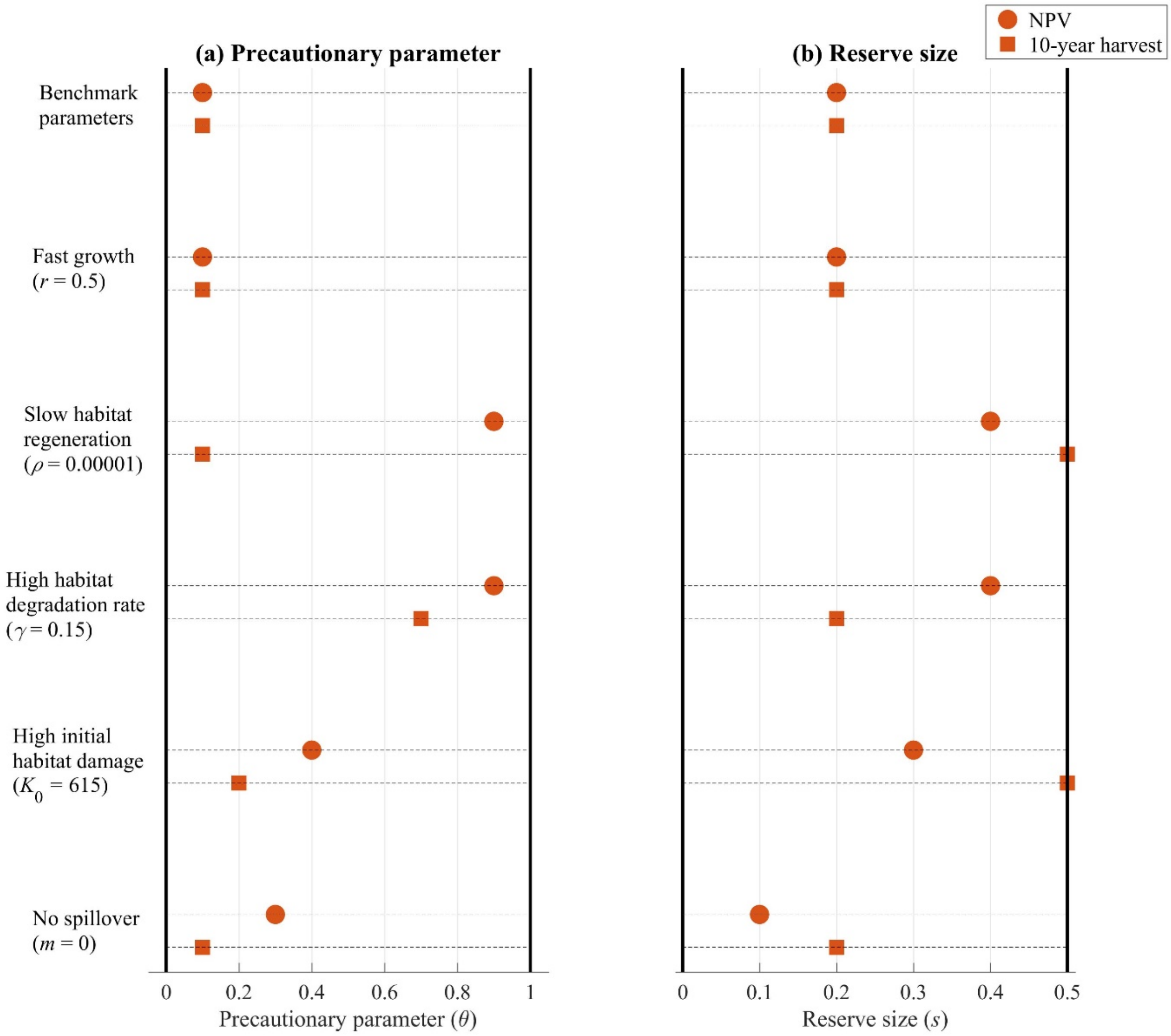
Regardless of whether habitat effects are present or not, less precaution is generally needed to maximise the NPV of the fishery and 10-year harvest as the mandated rebuilding constraint relaxes. However, the presence of habitat effects affects the way in which precaution ought to be applied in the rebuilding strategies. For example, where no habitat effects exist, precaution is exercised predominantly through the harvest control rule. For a given rebuilding constraint, no reserve or only 10% reserve is used to maximise the 10-year harvest and NPV of the fishery. Moreover, as the rebuilding constraint is extended, there is a transfer of precaution from the reserve to the harvest control rule. More specifically, with a 15-year rebuilding mandate, the strategy to maximise NPV is (0.2, 0.1); however, when the rebuilding constraint is relaxed to 20 years, the strategy changes to (0.3, 0). This shows that precaution that was exercised through the reserve is substituted by precaution applied through the harvest control rule as the rebuilding constraint becomes less restrictive.

In cases where the habitat can be degraded by fishing, on the other hand, a higher level of precaution is generally required than where there is no habitat effect, and precaution is now predominantly exercised through the reserve instead of through the harvest control rule. We also observe a transfer of precaution as the rebuilding constraint is extended, but now the precaution is transferred from the harvest control rule to the reserve. For example, when the rebuilding constraint is extended from 10 to 15 years, the reserve sizes for the strategies to maximise NPV and 10-year harvest increases to 0.2, while the required precaution exercised through the harvest control rule decreases.

2.3.5. Scenario Analysis

We examine the sensitivity of the optimal rebuilding strategies identified in the previous section to the value of key biological and environmental parameters by considering five different scenarios where (i) the growth rate of the stock is higher (r = 0.5); (ii) the habitat regeneration rate is slower (ρ = 0.0001); (iii) the degradation rate of the habitat is higher (γ = 0.15); (iv) the initial degradation of the habitat is more intense (K_0 = 615); and (v) there is no fish spillover (m = 0). Figure 7 shows combinations of the precautionary parameter and reserve size, (θ, s), that are compatible with the maximum NPV of the fishery and 10-year harvest for each of the five alternative scenarios. We confine our attention to the case of a rebuilding mandate of 20 years and where habitat effects exist. This is a case where, with the benchmark parameter values, there exists an optimal strategy, (0.1, 0.2), that simultaneously maximises both the NPV of the fishery and average 10-year harvest.

Figure 7. Changes in the optimal rebuilding strategy, (θ, s), that maximises the NPV of the fishery and 10-year average harvest with a 20-year rebuilding mandate. Benchmark parameters are reported in Table 1. Five alternative scenarios are considered: fast growth (r = 0.5); slow rate of habitat regeneration (ρ = 0.00001); high rate of habitat degradation (γ = 0.15); high initial damage of habitat (K_0 = 615); and no spillover (m = 0).



For the fast growth scenario, the optimal rebuilding strategy for both the NPV of the fishery and 10-year harvest remains unchanged, suggesting that there is no decrease or shift in precaution either through the harvest control rule or the reserve when the intrinsic growth rate is higher. Moreover, this is the only scenario for which the simultaneous maximisation of NPV and 10-year harvest still exists. For the other four scenarios considered it this paper, there is no unique combination of the precautionary parameter and reserve size that simultaneously maximise both the NPV of the fishery and average 10-year harvest, but only a single objective strategy exists. This suggests that, in these scenarios, trade-offs between the economic and socio-economic outcomes for the fishery is prevalent in meeting the 20-year rebuilding mandate.

A slower habitat regeneration rate or a higher degradation rate of the habitat has a pronounced impact on the single objective strategy that maximises either the NPV of the fishery or the average 10-year harvest. In these scenarios, that strengthen the habitat effect, the required level of precaution exercised through the harvest control rule and/or marine reserve increases. Similarly, if the habitat has initially been degraded further than originally assumed, the strategies to maximise NPV and 10-year harvest will require increased precaution through both the harvest control rule and marine reserve. The existence of spillover between the reserve and harvest populations also affects the way in which precaution is applied in single objective rebuilding strategies. That is, as the spillover of fish from the reserve to harvest population becomes non-existent, the strategy to maximise the NPV of the fishery substitutes the precaution that was exercised through the reserve by precaution applied through the harvest control rule; yet, a marine reserve will still form part of the optimal rebuilding strategy for maximising either NPV or average 10-year harvest.

2.4. Discussion

Global concern over the status of fisheries means that rebuilding depleted fish stocks, while taking into consideration the economic and social dimensions of fisheries, is a high priority for fisheries managers in many countries. The need to rebuild stocks, coupled with the imperative for precautionary management and a commitment to consider multiple objectives, therefore makes understanding how alternative mechanisms for controlling fishing mortality perform in stock recovery plans important. In this paper, we develop new insights into the outcomes of implementing the precautionary principle through two commonly used control mechanisms, harvest control rules and marine reserves, both individually and in concert. The relevance of our analysis for management is provided by comparing the ability of alternative stock rebuilding strategies to achieve the ultimate goal of rebuilding stocks in fisheries that are subject to fishing-induced habitat degradation.

Our results highlight the importance of accounting for the effect of fishing activity on habitat in the assessment of alternative stock rebuilding strategies by demonstrating the way in which a negative habitat effect may compromise the rebuilding objective. Our results show that, where habitat effects exist and precaution is exercised solely through the harvest control rule, the stock fails to rebuild for all but the highest level of precaution. This occurs because the habitat effect manifests as negative feedback on the fish stock biomass, undermining attempts to rebuild the biomass by limiting harvest. We find evidence of this negative feedback of habitat effects throughout our analysis, with the fish stock taking longer to rebuild, or failing to rebuild, regardless of whether only one or both control mechanisms are used in a stock recovery plan. Failure to include this negative feedback in the evaluation and design of stock recovery plans will therefore increase the risk of rebuilding objectives not being achieved.

Our results also confirm, and further strengthen, the case for considering marine reserves as part of the fisheries management toolbox, in the context of rebuilding depleted fish stocks. Importantly, we find that incorporating a reserve into a stock recovery plan will shorten the duration of the rebuilding period regardless of the presence of habitat effects. Moreover, in the case of a fishery with a habitat effect, the use of the reserve allows the rebuilding target to be achieved independent of the level of precaution exercised through the harvest control rule. These results highlight the dual role of the marine reserve in this context, where the reserve not only protects the biomass from harvest, but also effectively counteracts the negative feedback imposed on the fish stock biomass through the habitat-fishery linkage by preventing a decline in the carrying capacity (Armstrong, 2007). The coordination of the two control mechanisms thus permits the more efficient recovery of both the carrying capacity and biomass, promoting the shortened rebuilding period.

Furthermore, the substitutability of marine reserves and harvest control rules, demonstrated by Yamazaki et al., 2015, is maintained in our results, with the same stock rebuilding period being achieved for a range of combinations of the two control mechanisms. Our results extend this result further by showing that the ability for reserves to substitute the precaution exercised through the harvest control rule is greater with the habitat effect due to the hedge provided by the reserve against the negative feedback of habitat effects on the biomass. However, where the habitat is subject to an extremely high rate of damage, recovers slowly or has already suffered a high degree of loss prior to the implementation of the rebuilding strategy, the substitutability in the level of precaution exercised through different combinations of harvest control rule and reserve becomes restricted or disappears completely as the reserve is no longer capable of providing these hedging benefits.

The multi-faceted role of the marine reserve in a fishery with habitat effects is further demonstrated in our analysis of the trade-offs between fishery performance indicators. Our simulation consistently shows the trade-offs between biological, economic and socio-economic management objectives that are generally ubiquitous in fisheries management (Hilborn, 2007b; Mardle & Pascoe, 2002; Péreau et al., 2012). In this paper, the conflict between objectives in rebuilding depleted fish stocks is demonstrated by the fact that more precautious recovery plans that rely on both control mechanisms achieve stock rebuilding targets in a shorter timeframe than those that are less precautious, but at the expense of both the net present value of the fishery and the short-term harvest level. These same trade-offs are evident when precaution is implemented solely through one of the control mechanisms and regardless of whether fishing activity results in damage to the habitat or not.

In many fisheries management jurisdictions, however, managers must balance economic and socio-economic objectives in achieving the rebuilding target within a prescribed or mandated period of time. Our findings show that the nature of trade-offs between conflicting objectives is affected by whether habitat effect is a prevalent feature of the fishery and the way in which the precaution is implemented through the two mechanisms in a stock recovery plan. Notably, it is possible to identify a rebuilding strategy which meets a given rebuilding mandate with minimum trade-offs in economic and socio-economic objectives of fisheries management. We show that the likelihood of there being such a rebuilding strategy is greater when habitat effects exist and particularly if a relatively relaxed rebuilding timeframe is mandated. We further find that, where habitat effects exist, the exercise of precaution predominantly through a marine reserve is optimal even in the extreme case where there is no spillover from the reserve to the fishing grounds. These results reinforce our overall conclusion that a marine reserve serves as a hedge against negative feedback on the biomass through the habitat effect. Viewed differently, where the fishing-induced habitat damage is not a concern, the reserve's ability to provide these hedging benefits is reduced, as reflected in our results that shows that management precaution is optimally exercised mainly through the harvest control rule where there are either no habitat effects or the rate of habitat degradation is relatively low.

Overall, we show that, where a fishery is characterised by fishing-induced habitat damage, a stock rebuilding strategy which incorporates both harvest control rules and marine reserves will outperform a strategy which uses the two control mechanisms individually. Although the trade-off between the biological, economic, and socio-economic objectives of fisheries management is prominent in rebuilding depleted fish stocks, the extent of such a trade-off depends on the way in which precaution is implemented though different combinations of the two mechanisms. These results are important because they suggest that a failure to appropriately design precautionary fishery management may lead to a failure to rebuild fish stocks and may result in increased conflicts between different user groups in the marine environment.

Several caveats need to be noted when our results are applied and more work is needed to further explore the use of marine reserves and harvest control rule as effective means of rebuilding depleted fish stocks, particularly where the habitat-fishery linkage is inevitable. First, our modelling framework has not included age or size structure, larval dispersion or the use of marine reserves as protection of nursery or spawning grounds (De Leo & Micheli, 2015), which may mean the benefits to implementing a marine reserve as part of the stock rebuilding plan are greater than estimated here. Second, this study has not incorporated spatial characteristics of fisheries, such as the geographical distribution of fish stocks and habitat, as well as the behavioural responses of fishing fleets to the way in which the precaution is exercised through the two control mechanisms (Fulton et al., 2011; Pelletier & Mahévas, 2005; Smith & Wilen, 2003). Particularly, behavioural responses of resource users are considered as a major source of uncertainty in fisheries management (Fulton et al., 2011; Girardin et al., 2016), and failure to incorporate such information in the design of recovery plans may, therefore, result in unexpected management outcomes. Third, this study has focused on the net present value of the fishery and the short-term harvest as economic and socio-economic performance indicators. However, there is evidence that the impacts of marine reserves go beyond the commercial fishery and that they influence the values held by other resource user groups, such as recreational fisheries and marine-based tourism, and thereby the design of optimal reserve management depends on a range of use and non-use values of marine resources included in the decision process (Bhat, 2003; Grafton et al., 2011; Pascoe et al., 2014; Yamazaki et al., 2010).

One avenue for further research may be to assess community-wide impacts of alternative strategies for rebuilding depleted fish stocks.

Chapter 3: Allocation of harvest between users in a fishery with habitat effect

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3.1. Introduction

The importance of marine habitat quality to managing and maintaining sustainable fisheries is being increasingly recognised in fisheries legislation worldwide. [7] Causes of habitat degradation include natural phenomena such as hurricanes, land based causes such as coastal development, and fishing activity (Dayton et al., 1995; Jennings & Kaiser, 1998; Thrush & Dayton, 2002). The extent and nature of fishing impacts on the marine habitat vary according to the type of environment and the fishing gear used (Jennings & Revill, 2007). For example, marine environments such as coral reefs are particularly vulnerable to damage (Mangi & Roberts, 2006), and fishing gears such as bottom trawls can be particularly destructive (Hall-Spencer et al., 2002; Hiddink et al., 2017; Shephard et al., 2010; Thrush & Dayton, 2002). While the effects of such interactions may be mitigated directly through input controls and spatial reserves, the extent of habitat-fishery interactions and their consequences for fishery performance will also be influenced by the allocation of harvest across groups of fishery users.

Allocations of harvest grant a level of access to fish stocks to either an individual user or group of users and thereby determine the impact of fishing on the stock and habitat, and on how potential benefit from the resources are shared. Broadly speaking, the mechanisms to achieve allocations in fisheries may be: market-based such as auctions of harvest rights or transferable

[7] For example, the Magnuson-Stevens Fishery Conservation and Management Act s3(4) (USA), Fishery Management Act s3(1)(b), s14 (Australia) and the Common Fisheries Policy Reg.1380/2013 s4,11 (EU).

individual quota systems; administrative in nature such as allocations determined by fishery management or set by legislation; or some combination of the two. While administratively-based allocation processes may require managers to consider economic values of the resource for each user group (i.e. their allocative efficiency may not be as high as market-based transferable quotas (Arnason, 2012)), they have the advantage of enabling managers to consider and maintain control over the wider interests of the community when determining allocations (Morgan, 1995). Allocation processes based on administrative mechanisms have been used to distribute harvest within commercial fishing fleets, as well as allocate between sectors, including commercial, recreational and Indigenous in order to maintain the social and cultural value of a fishery (Crowe et al., 2013; Islam & Berkes, 2016; Lynham, 2014; Sutinen & Johnston, 2003). Administrative allocations have also been used to promote the economic development of local communities. An example of this are community development quotas (CDQs), which have been used in Alaskan fisheries as a way to guarantee catch for local fishing communities to encourage the economic development and sustainability of those communities (Fina, 2011; Ginter, 1995; Haynie, 2014; Holland & Ginter, 2001).

A key step underpinning the effectiveness of any harvest allocation is to decide the target level of biomass of the fishery and with this, the corresponding total catch for the fishery. While bioeconomic models allow managers to include information on the economic characteristics of fishing by different user groups when determining fishery-wide catch levels, they generally incorporate information on a single group of users only.[8] An understanding of how the optimal total catch and allocation over multiple user groups may be determined, and the consequences of deviating from this optimal allocation for commercial, social or cultural reasons, is highly

[8] There are exceptions to this, for example where different groups target different age groups of the same stock (Armstrong & Sumaila 2000), where cultural and commercial activities take place in the same location (Boncoeur et al., 2002), or where fishery resources are transboundary (Kronbak & Lindroos, 2007; Munro, 1979; Pintassilgo et al., 2015).

relevant for fishery managers who deal with competing stakeholders' claims to fishery resources, and who are also beholden to fishery management plans to maintain fish stock sustainability.

Knowing what economic and biological information needs to be included when calculating the target level of catch, biomass and allocation of catch shares across different user groups is important, particularly when management resources are scarce and failure to consider the impacts of fishing activity on marine habitats may inadvertantly undermine fish stock sustainability. In practice, however, calculation of biomass targets and catch levels rarely accounts explicitly for potentially damaging interactions between fishing and habitat. Moreover, the implications of these interactions for identifying the appropriate target level of catch and biomass is examined by only a small number of studies and remains generally poorly understood. The few exceptions include Foley et al., (2012) who found that the equilibrium biomass that maximizes the economic yield of the fishery may increase or decrease depending on how habitat-fishery interactions are modelled. More recently, Kahui et al., (2016) examined changes in economically optimal habitat and biomass in cold water coral environments and concluded that some environmental damage may be economically optimal, provided the environment is relatively healthy to begin with. Further to this, Armstrong et al., (2017) showed that non-use values of cold water corals affect this optimal biomass and habitat, by decreasing the optimal level of habitat destruction and increasing the optimal exploitation of biomass.

The overall aim of this paper is to extend this literature by exploring the optimal allocation of total catch between multiple user groups in a fishery where habitat-fisheries interactions are prevalent and how a deviation from this optimal harvest allocation impacts the fishery outcomes. We develop a bioeconomic model of a fishery where the carrying capacity of the biomass is impacted by fishing (i.e., habitat effect) and the regeneration of the habitat does not occur immediately but requires time. We consider two groups of users characterized by fishing technologies of differing environmental impact. We first analytically derive the marginal equilibrium conditions for which the net present value of the fishery is maximized where a single fish stock is targeted by two user groups. Using the conventional equilibrium conditions of Clark & Munro (1975) as a benchmark, we examine how the co-presence of habitat effects and multiple user groups influences the decision rule for optimal catch in the fishery. We then draw on the results of a simulated version of our model to explore the following three research questions. First, what is the extent of bias in the estimate of fishery-wide catch when habitat effects are ignored, and what are the consequences of the biased catch estimate for the fishery-wide profit and the biological outcomes, as measured by biomass and carrying capacity? Second, what are the impacts of habitat effects on the optimal allocation of harvest across different user groups, and how do the multi-user fishery outcomes differ from those that arise when only a single user group operates in the fishery? Third, how will fishery outcomes be affected when these catch shares are reallocated across user groups suboptimally?

The relevance of our findings for fisheries management is highlighted by contextualizing our simulations in two different management scenarios. In the first scenario, the fishery manager is directed to permit two user groups, characterized by different fishing technologies, within the same commercial fishing fleet to be used. This is relevant for fisheries where the optimal management choice may be to exclude the use of particular fishing technologies, but the opportunity cost of doing so is too great for fishers who have already invested in different technologies. In the second scenario, the fishery manager allocates harvest between two user groups who operate on different commercial scales. This is relevant for fisheries in which small fishing communities operate alongside large-scale commercial fishers, and fishery management wishes to ensure access to fish stocks to enable the economic viability of those groups.

3.2. Methods

3.2.1. Conceptual framework

In this section, we explore the economically optimal allocation of harvest in a fishery with habitat effects where two types of fishing gear are used: a highly efficient but environmentally destructive fishing gear (‘high impact technology’), and a less efficient gear which causes no environmental degradation (‘low impact technology’). The total catch of the fishery is the combined harvest of the high impact and low impact technologies. Similar to Kahui et al., (2016), the prevention or limitation of habitat damage occurs through the determination of the total catch in the fishery and an allocation of the catch between these two technologies, where a higher allocation to the low impact technology will mean less habitat damage.

The biomass dynamics are described as:

$$x_{t+1} = x_t + G(x_t, K_t) - h_t^H - h_t^L \tag{1}$$

where $G(.)$ is the growth function of the biomass x_t , h_t^H is harvest by the high impact technology and h_t^L is harvest by the low impact technology. This paper incorporates the effect of fishing-induced habitat changes on fish biomass through the carrying capacity of the population, K_t.[9] We assume that the carrying capacity increases when the habitat recovers and decreases when the habitat is damaged due to fishing (Elliott & Cutts, 2004). Following Upton and Sutinen, (2005) and Udumyan et al., (2010), the carrying capacity dynamics are described as:

$$K_{t+1} = K_t + H(K_t) - D(h_t^H, K_t) \tag{2}$$

[9] Foley et al., (2012) provides a classification of types of habitat in bioeconomic modelling; according to this classification, this model represents a type of environment where habitat improvements could increase carrying capacity through improved nursery grounds, refuge from predators or increased nutrient supply.

where $H(.)$ is the growth function of the population carrying capacity and $D(.)$ is the damage function.

Net profit at time t is given by:

$$\pi_t = \pi(h_t^H, h_t^L, x_t) = TR(h_t^H, h_t^L) - TC(h_t^H, h_t^L, x_t) \quad (3)$$

The total revenue to the fishery, *TR*, is an increasing function of the harvest of both fishing technologies, i.e., $\partial TR / \partial h_t^j > 0$, $j = H, L$. The total cost of fishing, *TC*, is a function of the harvest of the two technologies and the fish stock biomass. We assume that harvesting costs decrease as biomass increases and that the marginal cost of harvesting decreases as the abundance of the fish population increases, i.e., $\partial TC / \partial h_t^j > 0$, $\partial TC / \partial x_t < 0$ and $\partial^2 TC / \left[\partial h_t^j \partial x_t\right] < 0$, $j = H,L$. We further assume that the low impact technology is less efficient at harvesting than the high impact technology, such that the low impact fishing technology has a higher marginal cost of fishing than the high impact fishing technology, i.e., $\partial TC / \partial h_t^H < \partial TC / \partial h_t^L$ for given x_t.

Given equation (3), the net present value (NPV) of the fishery is given as:

$$NPV = \sum_{t=0}^{\infty} \frac{1}{(1+\delta)^t} \pi(h_t^H, h_t^L, x_t) \quad (4)$$

where $\delta \in [0,1]$ is the discount rate.

3.2.2. Marginal equilibrium conditions with two technologies and habitat effects

We derive the marginal equilibrium conditions where the habitat effect exists and the fishery manager controls habitat impacts through the allocation of catch shares between fishing technologies. Given the NPV of the fishery in (4), the optimization problem is specified as:

$$\max_{\{h_t^H, h_t^L\}_{t=0}^{\infty}} \sum_{t=0}^{\infty} \frac{1}{(1+\delta)^t} \pi(h_t^H, h_t^L, x_t) \tag{5}$$

subject to the biomass dynamics (1), carrying capacity dynamics (2), and the initial conditions h_0^H, h_0^L, x_0, and K_0. The Lagrangian function for this problem is defined as:

$$\begin{aligned}\mathcal{L}\left(h_t^H, h_t^L, x_t, K_t, \mu_t^x, \mu_t^K\right) = \sum_{t=0}^{\infty} \frac{1}{(1+\delta)^t} \Big[& \pi(h_t^H, h_t^L, x_t) + \mu_t^x (x_t + G(x_t, K_t) - h_t^H - h_t^L - x_{t+1}) \\ & + \mu_t^K (K_t + H(K_t) - D(h_t^H, K_t) - K_{t+1}) \Big]\end{aligned} \tag{6}$$

The Lagrangian multiplier μ_t^x gives the marginal value of the biomass at time *t*, while the multiplier μ_t^K shows the marginal value of the carrying capacity at time *t*. The terms μ_t^x and μ_t^K therefore represent the shadow price (that is, the value imputed from future productivity) of the fish stock biomass and carrying capacity.

The first order conditions (FOC) for maximizing the Lagrangian function (6) are given by:[10]

$$\begin{aligned}\pi_{h_t^H} &= \mu_t^x + \mu_t^K D_{h_t^H} \\ \pi_{h_t^L} &= \mu_t^x \\ \pi_{x_{t+1}} &= (1+\delta)\mu_t^x - \mu_{t+1}^x \left(1 + G_{x_{t+1}}\right) \\ \mu_{t+1}^x G_{K_{t+1}} &= (1+\delta)\mu_t^K - \mu_{t+1}^K \left(1 + H_{K_{t+1}} - D_{K_{t+1}}\right)\end{aligned}$$

along with the biomass and carrying capacity dynamics in (1) and (2).

The first FOC implies that the marginal benefit to profit associated with an extra unit of harvest by the high impact technology will be equal to the marginal cost associated with lost future productivity for biomass and carrying capacity, with the carrying capacity term weighted by the damage rate. The second FOC gives the same condition for the low impact technology, but

[10] A function with a subscript denotes a partial derivative with respect to the subscripted variable other than *t*.

since the low impact technology is assumed to cause no habitat damage, the marginal cost comprises the value of lost future productivity of biomass only. The third FOC suggests that the benefit to profit associated with a marginal increase in biomass will be equal to the marginal cost, comprised of the present value of the shadow price of the fish stock plus the future shadow price, weighted by future stock growth. The fourth FOC indicates that the benefit to biomass from a marginal increase in the carrying capacity is equal to the marginal cost, comprised of the present value of the shadow price of the carrying capacity plus the future shadow price, weighted by future net growth in carrying capacity.

At the steady state where $x_{t+1} = x_t = \bar{x}$, $K_{t+1} = K_t = \bar{K}$ and $h_{t+1}^j = h_t^j = \bar{h}^j$, j=H, L, we find the marginal equilibrium conditions for low and high impact technologies as:

$$\delta = \frac{\pi_{\bar{x}}}{\pi_{\bar{h}^L}} + G_{\bar{x}} \tag{7}$$

$$\frac{\pi_{\bar{x}}}{\pi_{\bar{h}^H}} = \frac{(\delta - G_{\bar{x}})(\delta - H_{\bar{K}} + D_{\bar{K}})}{(\delta - H_{\bar{K}} + D_{\bar{K}}) + G_{\bar{K}} D_{\bar{h}^H}} \tag{8}$$

Equation (7) shows the fundamental equation of renewable resources as described in Clark and Munro (1975), when the fish is harvested by a fishing technology that does not cause habitat damage. This equation demonstrates that the economically optimal fish stock biomass can be found where the marginal opportunity cost of delaying harvest (which is equal to the discount rate) is equal to the marginal benefit of delaying harvest. The marginal benefit consists of the direct growth benefit that accrues from allowing the biomass to increase $(G_{\bar{x}})$, plus the decrease in harvesting costs due to the increased density of biomass (the marginal stock effect $\pi_{\bar{x}} / \pi_{\bar{h}^L}$).

Equation (8) is analogous to this fundamental equation for a fishing technology which causes habitat degradation. The left hand side of equation (8) describes the marginal stock effect for the high impact technology. The right hand side of equation (8) consists of three different terms: the net growth benefit $(\delta - G_{\bar{x}})$, the net habitat benefit $(\delta - H_{\bar{K}} + D_{\bar{K}})$ and the interactive damage effect $G_{\bar{K}} D_{\bar{h}^H}$. The net growth benefit consists of the discount rate and the direct benefit of delaying harvest to stock growth, and so shows the marginal net opportunity cost to the stock of delaying harvest. The net habitat benefit consists of the discount rate, the direct benefit of delaying harvest for carrying capacity growth less the habitat damage, and is therefore the marginal net opportunity cost to the carrying capacity of delaying harvest. The interactive damage effect consists of the marginal change in stock growth caused by a change in carrying capacity and the marginal damage to carrying capacity caused by the high impact technology. Put together, this interactive term describes the damage caused to the biomass and carrying capacity due to the habitat effect.

Solving the system of equations (7) and (8), along with the biomass and carrying capacity dynamics at the steady state, yields the biomass, carrying capacity, harvest and catch shares at the maximum economic yield. These optimality conditions cannot be solved explicitly, and so the numerical solutions are simulated using the dynamics and parameter values specified below.

3.3. Numerical model

3.3.1. Functional specifications

The growth functions of the biomass and carrying capacity are specified as:

$$G\left(x_t, K_t\right) = rx_t\left(1 - \frac{x_t}{K_t}\right) \text{ and } H(K_t) = \rho K_t\left(1 - \frac{K_t}{K_{MAX}}\right) \tag{9}$$

where r is the intrinsic growth rate of the population, ρ is the intrinsic regeneration rate of the population carrying capacity, and K_{MAX} is the upper limit on the population carrying capacity. The habitat damage function is specified as:

$$D(h_t^H, K_t) = \gamma h_t^H K_t \tag{10}$$

where γ is the rate of degradation caused by the high impact technology.

Net profit at time t is given as:

$$\pi(h_t^H, h_t^L, x_t) = ph_t^T - \frac{c^H h_t^H}{x_t} - \frac{c^L h_t^L}{x_t} \tag{11}$$

where p is the price of fish, $h_t^T = h_t^H + h_t^L$ is the total catch in the fishery, and c^H and c^L are cost parameters for the high and low impact fishing technologies.

3.3.2. Parameter values

The benchmark parameter values used in the simulations are presented in Table 1. The bioeconomic model is simulated with the estimated parameter values reported in Yamazaki et al., (2015) for the South Georgian Patagonian toothfish fishery. This fishery is a single-species fishery, where the target species is harvested by longline fishing gear, although historically trawlers have also operated in the area. The purpose of this analysis is not to provide an empirical evaluation of this specific fishery, but to develop insights into how the optimal catch and allocation of harvest between multiple user groups in a fishery where habitat effects exist ought to be determined, and how suboptimal catch shares impact fishery performance.

Table 1. Parameter values

Parameter	Description	Value
p	Price of fish (USD per tonne)†	9131
c^H	Cost parameter (high impact technology)†	2.62E+08
c^L	Cost parameter (low impact technology)††	3.14E+08
δ	Time discount rate†	0.05
r	Intrinsic growth rate†	0.12
K^{MAX}	Maximum carrying capacity (tonnes) †	109,225
γ	Damage rate of habitat††	3.00E-06
ρ	Regeneration rate of habitat††	[0.01, 0.2]

† The parameter values are taken from Yamazaki et al. (2015)

†† The parameter values are calculated or set by authors.

We assume that the cost parameter for the low impact technology is 20% higher than that for the high impact technology, to reflect the lower efficiency of this fleet. The regeneration rate ρ is given a range between 0.01 and 0.2.[11] We test the sensitivity of the results to changes in the damage rate of the habitat, the relative fishing costs of the two technologies and intrinsic growth rate in Section 4.5.

3.4. Simulation results

3.4.1. Accounting for habitat effect in the calculation of optimal catch

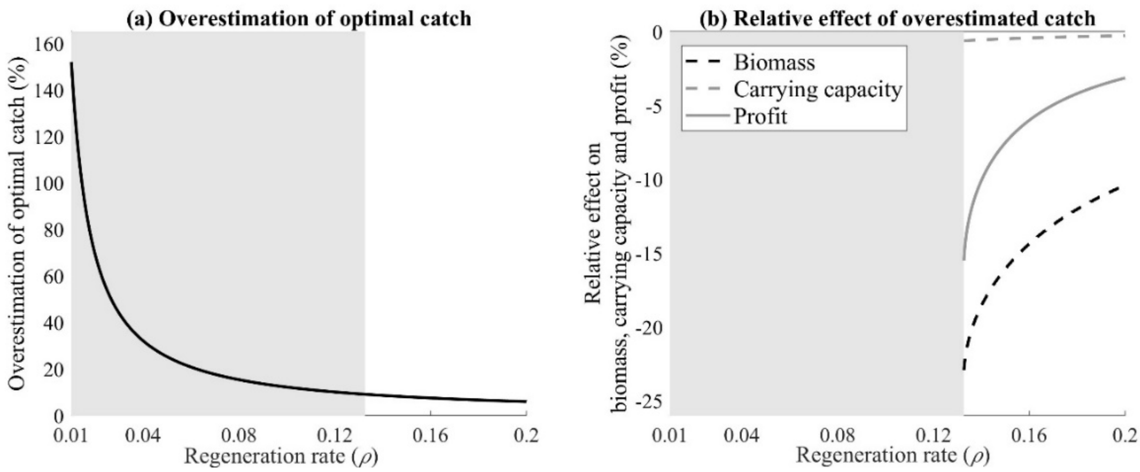
We first examine the consequences of failing to account for habitat effects when determining the catch level that maximizes the NPV of the fishery (i.e., optimal catch) where fishing-

[11] While longline fishing gear has been shown to have a relatively low environmental impact (Pham et al., 2014), the impacts of this fishing gear on the South Georgian fishery, and knowledge of recovery rates of the South Georgian benthic habitat generally, is lacking (Hogg et al., 2016). In this model, a regeneration rate of 0.01 corresponds to a time period of 295 years for the carrying capacity to go from half its maximum, to regenerating completely, assuming no further damage is taking place. A rate of 0.2 corresponds to a time period of 15 years.

induced habitat damage exists. In this analysis, we consider only the high impact technology for which the damage rate (γ) is positive. We exclude the low impact technology from this analysis since we assume that the low impact technology causes no habitat damage and $c^L > c^H$. It would therefore be irrelevant to the results to include fishing by this technology, as it will not be economically optimal to make an allocation to a low impact technology when habitat effects are omitted in the determination of the optimal catch. To show the consequences of failing to account for habitat effects, we calculate the relative difference in equilibrium biomass, carrying capacity and profit of the fishery between two cases: first, where the optimal catch is calculated without taking into account the carrying capacity dynamics (i.e., habitat effects) of equation (2), and second, where habitat effects are included in the calculation of the optimal catch. In this way, we assess the consequences of failing to internalize the habitat effect via the bioeconomic model. In the next section, we examine fishery outcomes where both multiple technologies and habitat effects are included.

When habitat effects are ignored, the optimal catch will be overestimated for every regeneration rate considered in this paper (Figure 1a). In other words, the calculated catch without accounting for habitat effects is higher than the catch which accounts for habitat effects of fishing. The extent to which the optimal catch is overestimated is particularly high (> 100%) in a slow-growing environment ($\rho < 0.02$), but decreases as the regeneration rate increases. For example, in the slowest regenerating environment ($\rho = 0.01$), the optimal catch will be overestimated by 151%, but in the fastest growing environment ($\rho = 0.2$), the optimal catch is overestimated by only 6%.

Figures 1: (a) Overestimation of the optimal catch as a result of failing to account for habitat effects. (b) Relative effect of overestimated catch on equilibrium biomass, carrying capacity and profit of the fishery. The baseline case is the biomass, carrying capacity and profit when habitat effects are included in the calculation of the optimal catch. The range of habitat regeneration rate, below which a failure to account for habitat effects results in the stock collapse, is shown by the shaded area.

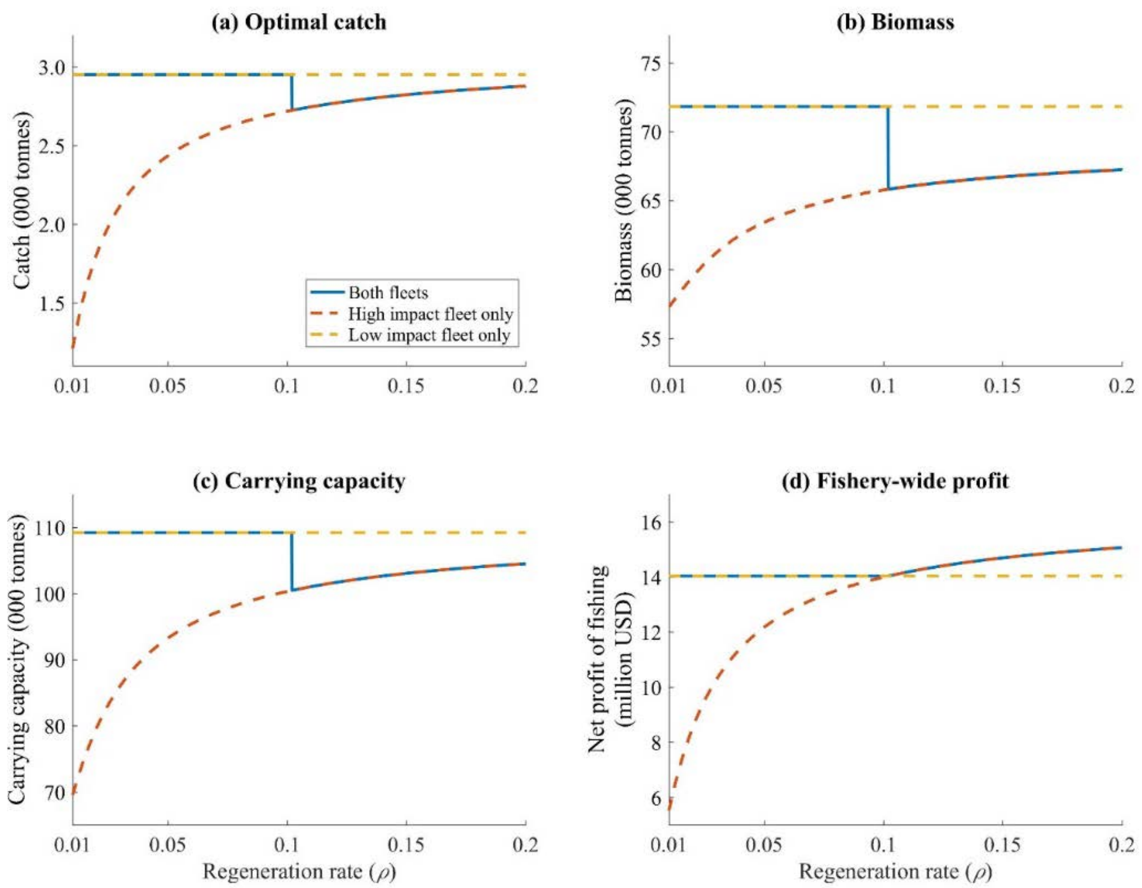


Since the level of overestimation decreases exponentially as the rate of habitat regeneration increases, the extent to which the optimal catch is overestimated is relatively low and stable in a range of fast-growing environments. For example, the level of overestimation is between 5 and 20% for regeneration rates between 0.06 and 0.2. Within this range, however, there is a threshold (ρ^*) such that for regeneration rates less than ρ^*, the failure to account for habitat effects leads to collapse of the fish stock. Stock collapse is depicted by the shaded area in Figure 1 for values of $\rho < \rho^* = 0.13$. This threshold for stock collapse exists because, although the direct negative effects of the overestimated catch on the carrying capacity are small relative to the effects on the biomass for low regeneration rates, such moderate habitat effects coupled with the overexploitation of biomass can lead to collapse of the fish stock. However, as the regeneration rate increases the consequences of overestimation for fishery performance are mitigated, and so we see the relative effects decrease towards zero (Figure 1b).

3.4.2. Fishery outcomes with habitat effect and multiple technologies

In this section, we explore the implication of habitat effects and multiple fishing technologies on the optimal allocation of catch shares across different user groups and the resulting fishery outcomes in terms of biomass, carrying capacity and fishery-wide profit. Our results are given real world context by considering the case of a commercial fishery comprising two fleets characterised by their fishing technologies, which are referred to as the *high impact fleet* and the *low impact fleet*, respectively. We use the analytical solutions shown in equations (7) and (8) to calculate the optimal catch, biomass and carrying capacity of the fishery, when the NPV of the fishery is maximised over both fleets. These solutions also give the optimal catch share of each of the two fleets. We compare these multi-fleet outcomes with those that arise when only a single fleet operates in the fishery (i.e., only one fleet is considered in the calculation of the optimal catch).

Figures 2: Optimal catch (a), biomass (b), carrying capacity (c) and fishery-wide profit (d) for different regeneration rates. The solid blue line shows the outcome where the NPV of the fishery is maximized over both fleets, the red dotted line shows the outcome where the NPV of the fishery is maximized when only the high impact fleet operates, and the yellow dotted line shows the outcome where the NPV of the fishery is maximized when only the low impact fleet operates.



When the NPV of the fishery is maximized over both fleets, the entire catch in the fishery is optimally harvested by a single fleet only, for all regeneration rates considered in this paper (Figure 2a). The equilibrium levels of biomass, carrying capacity and profit thus all coincide with the outcomes when only a single fleet is considered in calculating the optimal catch (Figure 2b-d). In other words, when determining the catch share between the two fleets with different fishing technologies, it is optimal to make no allocation of harvest to either the high or low impact fleet. Specifically, for a regeneration rate less than 0.1, the multi-fleet case and the low impact single-fleet case yield the same optimal outcome. By contrast, for regeneration rates in excess of 0.1, the optimal outcome in the multi-fleet case is the same as maximizing the NPV of the high impact fleet only.[12]

These results also imply that the use of the high impact technology in a slow-growing environment (ρ < 0.1) or the low impact technology in a fast-growing environment (ρ > 0.1) leads to a deviation from the multi-fleet optimum. For example, when only the high impact fleet is considered in a slow-growing environment, the optimal catch, and the resulting biomass, carrying capacity and profit, are all relatively low. By contrast, although the use of low impact technology is not optimal in a fast-growing environment (ρ > 0.1), the catch, biomass and carrying capacity that arise with only the low impact fleet in these environments are greater than those with the high impact fleet for any regeneration rate. However, for regeneration rates greater than 0.1, the overall profit of the fishery where only the low impact technology is used is lower than where only the high impact technology is used.

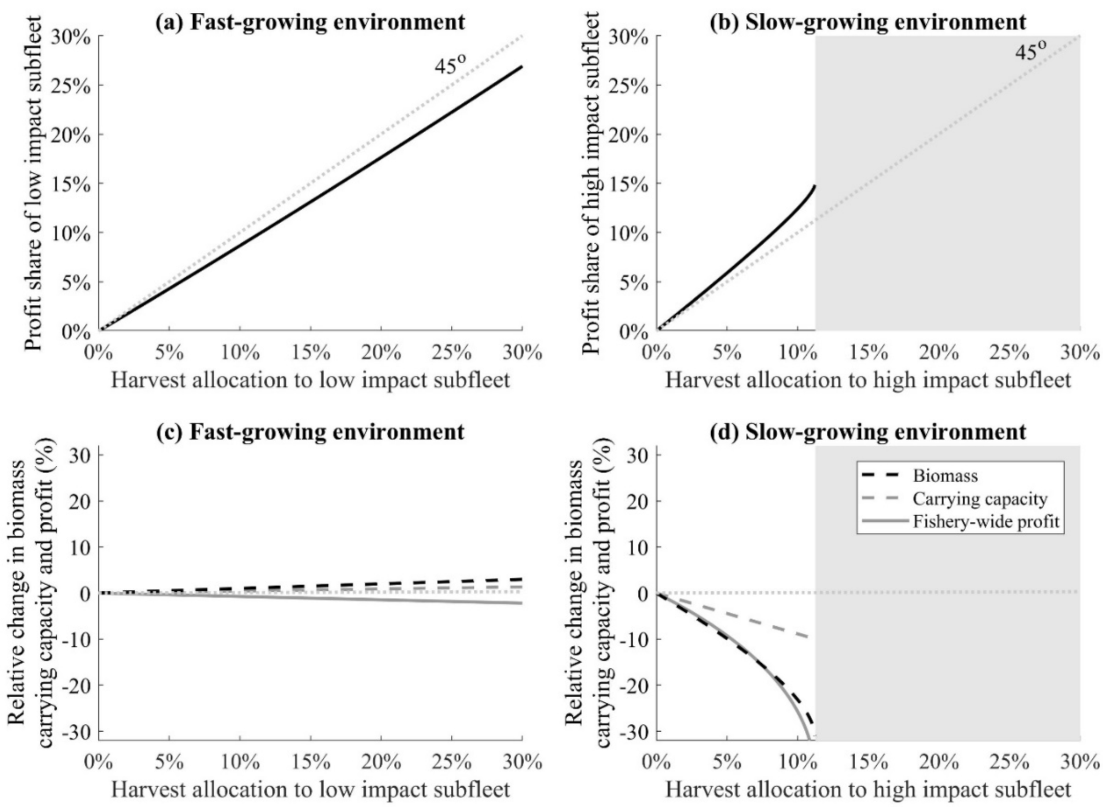
3.4.3. Effects of suboptimal catch shares

The results above show that the maximization of the NPV of the fishery over multiple user groups with different fishing technologies requires the catch share of one of the user groups to be set at zero. In this section, we explore how fishery outcomes change when the catch is suboptimally allocated to ensure both fleets receive a positive catch share. We consider two cases: first, a suboptimal harvest allocation to the low impact fleet is made in the fast-growing environment (ρ = 0.2) where the optimal catch is achieved with the high impact fleet only, and

[12] When maximising profit, there is a balancing between the relatively high marginal cost of fishing by low impact fleet and the fishing-induced habitat damage by high impact fleet. For slow regeneration rates, the balance favours the habitat damage, and for fast regeneration rates, the balance favours cost, such that one fleet is excluded when maximising profit.

second, a share of catch is allocated to the high impact fleet in the slow-growing environment (ρ =0.01) where the optimal catch is achieved with the low impact fleet only.

Figures 3. Changes in the profit share of low impact fleet in the fast-growing environment, ρ = 0.2, (a) and high impact fleet in the slow-growing environment, ρ = 0.01, (b) when the harvest is suboptimally allocated. Relative effect of the harvest allocation on biomass, carrying capacity and fishery-wide profit in the fast-growing (c) and slow-growing environment (d). The baseline case for panels (c) and (d) is where the optimal allocation of catch shares is made in the fast-growing and slow-growing environment, respectively.



As the share of catch allocated to the low impact fleet increases in the fast-growing environment, this fleet's share of profit also increases (Figure 3a). Moreover, as progressively more catch is allocated to the low impact, but less efficient, fleet, the overall profit of the fishery moderately declines (Figure 3c). Despite the declining overall profits of the fishery, the allocation to the low impact fleet increases both the biomass and the carrying capacity of the fishery, reflecting the zero damage rate of this fleet. These results mean that allocations that deviate from the optimal catch share of the two fleets in the fast-growing environment create a trade-off between improving the biological outcomes for the fishery (i.e. biomass and carrying capacity) and sacrificing the fishery-wide profit. However, the extent of such tradeoffs is weak; for example, the decision to allocate 30% of the harvest to the low impact fleet results in less than a 3% change in overall profit, biomass and carrying capacity.

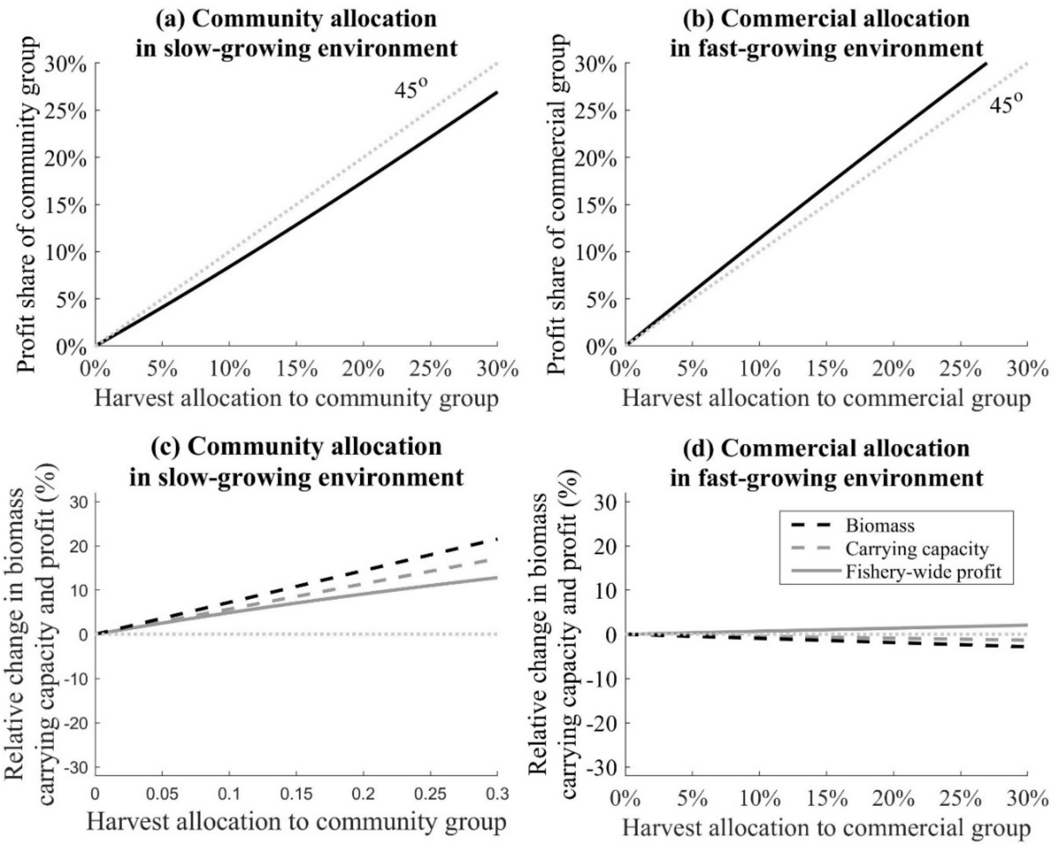
For the case of the slow-growing environment (ρ =0.01) in which allocation of the entire catch to the low impact fleet maximizes the NPV of the fishery, allocations of 11% or less of the catch to the high impact fleet increase the profit share of this fleet at an increasing rate (Figure 3c). However, the increase in the profit share of the high impact fleet is achieved at the expense of diminished fishery performance across both biological outcomes (i.e., biomass and carrying capacity) and fishery-wide profit (Figure 3d). The extent of the change in fishery outcomes due to suboptimal harvest allocations is greater when made in favor of the high impact fleet in the slow-growing environment than when the catch is allocated to the low impact fleet in the fast-growing environment. For example, when the allocation to the high impact fleet is 10%, overall fishery profit and biomass decline by approximately 25%, and carrying capacity declines by 9%. By contrast, an allocation of the same magnitude in the fast-growing environment results in a less than 1% change in these outcomes. The consequence of deviating from the optimal catch share in the slow-growing environment may also be extreme. That is, the allocation of more than 10% of the high impact fleet results in stock collapse, shown by the shaded area in Figure 3 (b,d). This collapse reflects the inability of habitats with low regeneration rates to support a higher proportion of high impact, destructive fishing activity.

3.4.4. Using catch shares to maintain multiple user groups in the fishery

In this section, we examine how harvest allocations may affect fishery performance where the (suboptimal) total catch of the fishery is calculated based on a single use group instead of multiple groups. Our results in this section are given real world context by considering a situation in which the fishery-wide catch and biomass target are based on the assumption of a single user group, but catch shares are then allocated across multiple groups. This type of situation may arise in fisheries where secondary allocations are used to grant or maintain access to fish stocks for either larger scale commercial fishers, or community groups who fish on a smaller commercial scale (Grieve, 2009).

For the purpose of our analysis, we characterize small-scale commercial users as the low impact technology users and refer to them as the *community group*; and the high impact technology users as the large-scale commercial users which are referred to as the *commercial group*. We also limit our exploration to two different circumstances: first, where the fishery-wide catch is determined based solely on the commercial group in the slow-growing environment ($\rho = 0.01$), and a proportion of the catch is then allocated to the community group, and second, where the total catch in the fishery is based solely on the community group in the fast-growing environment ($\rho = 0.2$), and the harvest is then proportionally allocated to the commercial group.

Figures 4. Changes in the profit share of community group in the slow-growing environment, ρ = 0.01, (a) and commercial group in the fast-growing environment, ρ = 0.2, (b) when catch shares are used to maintain multiple user groups. Relative effect of the harvest allocation on biomass, carrying capacity and fishery-wide profit in the slow-growing (c) and fast-growing environment (b). Baseline case for panel (c) is when the commercial group is the sole operator in the slow-growing environment. Baseline case for panel (d) is when the community group is the sole operator in the fast-growing environment.



As the community group's allocation increases in the slow-growing environment, a larger proportion of profits accrue to the less efficient community group (Figure 4a). Despite this, the overall profit of the fishery increases (Figure 4c). For example, when community fishers are allocated 30% of the total harvest, fishery-wide profit increases by 13%. This is because the community group causes no environmental degradation, which in the slow-growing environment, means that higher profits are earned for the same catch, due to improvements in biomass and carrying capacity. This suggests that, in the environment where habitat is slow to recover from fishing-induced damage, community allocations have the potential to improve both the biological outcomes and fishery-wide profit.

Similarly to the slow-growing environment, the profit share of the commercial group and the overall profits of the fishery both increase in the fast-growing environment with increasing allocations of harvest to commercial users (Figure 4b,d). Despite more habitat damage, higher overall fishery profits occur because the commercial users are assumed to be more efficient at harvesting fish, decreasing the total cost of harvesting. In the fast-growing environment, the high regeneration rate of habitat is able to support the higher proportions of environmentally damaging fishing activity. However, the gains in profit are substantially lower compared to those that occur in the slow-growing environment. For example, an allocation of 20% of the total harvest to the secondary user group will result in 1.4% and 9% increase in overall profits, in the fast- and slow-growing environments, respectively. Moreover, the modest increase in overall profits in the fast-growing environment is achieved at the expense of a slight decline in both biomass and carrying capacity (Figure 4d), suggesting a weak trade-off between the economic performance and biological outcomes of the fishery due to the increase in habitat degradation that arises from use of the high impact fishing technology.

3.4.5. Sensitivity analysis

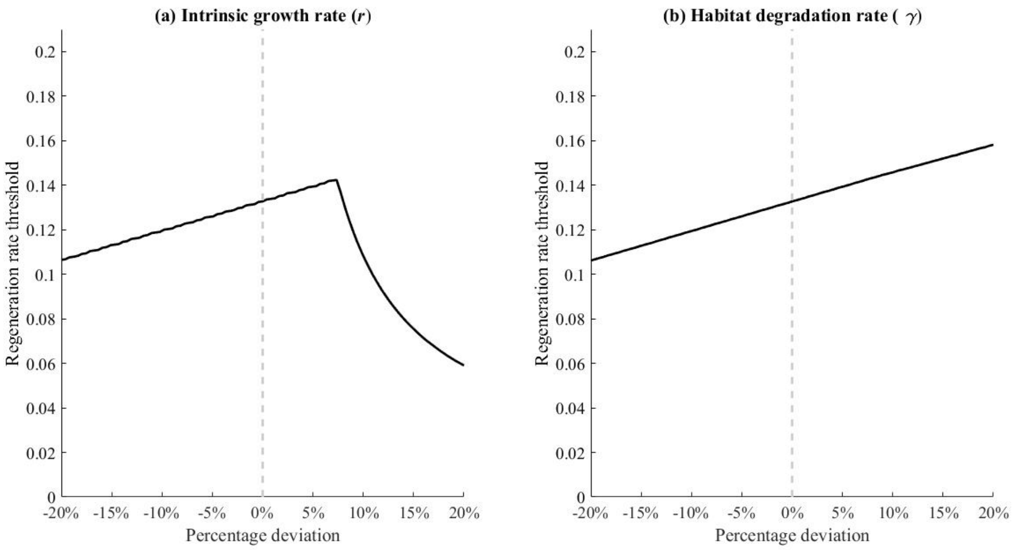
In this section, we examine the sensitivity of our main simulation results to changes in the value of key biological and economic parameters by considering a range of intrinsic growth rates (r), habitat damage rates (γ) and relative fishing costs (c^L/c^H). In particular, we explore how changes in these parameters affect: (i) the potential for stock collapse due to the omission of habitat effects in the determination of the optimal catch for the fishery; (ii) the optimal allocation of harvest between fleets; and (iii) the potential for stock collapse through harvest allocations to the high impact fleet in a slow-growing environment.

3.4.5.1. Stock collapse through a failure to account for habitat effects

In Section 4.1, we found that stock collapse could occur if habitat effects were not incorporated into the determination of the optimal catch for the fishery. Stock collapse occurred in slow-growing environments where the habitat regeneration rate was less than a threshold value (ρ^*) equal to 0.13, because the optimal catch for the fishery is overestimated (Figure 1a). Figures 5 shows the change in ρ^* for ±20% deviations from the baseline values of the intrinsic growth rate (r) and the damage rate (γ).[13] As the intrinsic growth rate decreases below the baseline value, ρ^* decreases and the range of habitat regeneration rates at which stock collapse occurs becomes smaller (Figure 5a). For example, at the baseline value of r, the stock collapse will occur for every value of $\rho < \rho^* = 0.13$, whereas when the growth rate is 20% lower than the baseline value, this stock collapse will occur only where $\rho < \rho^* = 0.10$. This means that, for a slow-growing stock, it is only in the slowest growing environments that stock collapse will occur as a result of omitting habitat effects in the calculation of the optimal catch for the fishery.

[13] Only changes in these parameter values are considered here because variations in cost have no impact where a single fleet with habitat effects is considered in the calculation of the optimal catch.

Figures 5. Sensitivity of the threshold of the habitat regeneration rate (ρ*) at which stock collapse occurs as a result of an overestimation of the optimal catch due to omitting habitat effects, for ±20% deviations in the intrinsic growth rate (a) and the habitat damage rate (b). The 0% marker indicates the baseline value of both parameters.



As the intrinsic growth rate increases above the baseline, the threshold of habitat regeneration rate continues to increase. However, there is a tipping point beyond which this relationship is negative, such that as r increases further ρ* declines. For example, when the growth rate is 20% greater than the baseline value, stock collapse will occur only where $\rho < \rho^* = 0.06$. This means that when the intrinsic growth rate of the stock is relatively fast, the fishery does not collapse by failing to account for the habitat effect, unless the habitat is extremely slow growing.

In summary, the parabolic nature of the relationship between the intrinsic growth rate and the threshold of habitat regeneration rate means that the range of lowered threshold of habitat regeneration rate is observed for either extremely slow-growing or extremely fast-growing stock. In the case of a relatively low intrinsic growth rate, the extent to which the optimal harvest is overestimated is also relatively low as the optimal catch is set precautionary regardless of the habitat effect; and therefore the omission of the habitat effect from the determination of the optimal catch has little effect. The extent of overestimation initially increases as the intrinsic growth rate increases, resulting in an increase in the likelihood of stock collapse even in a fast-growing environment (e.g., ρ = 0.14). However, sufficiently fast-growing stocks are resilient to the omission of the habitat effect, and so the range of habitats for which for stock collapse occurs is narrowed.

In contrast, the relationship between the threshold of habitat regeneration rate and the damage rate is monotonic and positive (Figure 5b). When the habitat damage rate is lower than the baseline value, the threshold at which stock collapse occurs is lower, suggesting that stock collapse will occur only where the habitat regeneration is relatively low. However, the failure to account for habitat effects in calculating the optimal catch for the fishery becomes more severe as the damage rate increases, with the fish stock collapsing even for higher values of the habitat regeneration rate.

3.4.5.2. Optimal allocation between fleets in a commercial fishery

Our results in Section 4.2 suggest that it is economically optimal to make no allocation of harvest to either the high impact or low impact fleet and which fleet receives a positive catch share depends on the habitat regeneration rate. To explore the sensitivity of this result, we run a Monte Carlo simulation over ranges of the intrinsic growth rate $(r \pm 30\%)$, habitat damage rate $(\gamma \pm 30\%)$, relative fishing costs $(\mathrm{c}^{L}/\mathrm{c}^{H} \in [1,1.4])$ and habitat regeneration rate $(\rho \in [0.01, 0.2])$, for 1000 iterations. We use a uniform distribution to draw a value of these parameters randomly from the range, and then find the optimal allocation of catch shares between the high impact and low impact fleets.

The results from the sensitivity analysis show that close to 80% of the time, maximizing the overall profit of the fishery requires the catch share of either the high impact or low impact fleet to be set at zero. More particularly, 41% of the time only the high impact fleet receives a share, 37% of the time only the low impact fleet receives a share, and 22% of the time both fleets receive a positive share of the catch. This suggests that more often than not, the catch will be allocated entirely to either the high impact or low impact fleet, and so the majority of optimal allocations considered in this paper result in the exclusion of users from the fishery.

We also examine the sensitivity of the optimal harvest allocation to changes in the intrinsic growth rate, habitat damage rate, habitat regeneration rate and relative fishing costs by estimating a generalized linear model (GLM) with a logit link and binomial family (Table 2). Given the nonlinear relationship between the intrinsic growth rate and stock collapse observed in Section 4.5.1, a squared intrinsic growth rate term was included to account for the possibility of this relationship also existing between the optimal allocation and the intrinsic growth rate. The estimation results show that the coefficient for the intrinsic growth rate is -0.042 and the coefficient for the squared term is 0.0005, suggesting that the relationship between the intrinsic growth rate and the optimal allocation is convex.[14] These results imply that the optimal proportion of harvest allocated to the high impact fleet initially decreases as the intrinsic growth rate increases. However, as the growth rate further increases, this relationship reverses, so that the optimal allocation to the high impact fleet proportionally increases with an increase in the growth rate. This convex relationship exists for the same reason as for the parabolic relationship between the growth rate and the threshold of habitat regeneration rate (Section 4.5.1). For low stock growth rates, the total catch for the fishery is set precautionary, and so the proportion of catch allocated to the high impact feet is relatively high. By contrast, for high stock growth rates, the fish stock become resilient to the habitat degradation caused by fishing, and so a larger allocation to the high impact fleet is permitted.

[14] The statistical insignificance of the squared term may be due to the number of Monte Carlo observations or the range of parameter values considered in the simulation.

As one would expect, if the habitat damage rate increases or the habitat regeneration rate decreases, the optimal proportion of harvest allocated to the high impact fleet will decrease. We further find that, when the relative cost of fishing to the low impact fleet increases by 1%, the optimal allocation of harvest to the high impact fleet increases by 0.41%. The relative size of estimated coefficients also suggest that the relative cost of fishing has a relatively larger effect on the optimal allocation of harvest between fleets, compared to the biological parameters.

Table 2. GLM regression results

Dependent variable: optimal proportion of harvest allocated to high impact fleet.

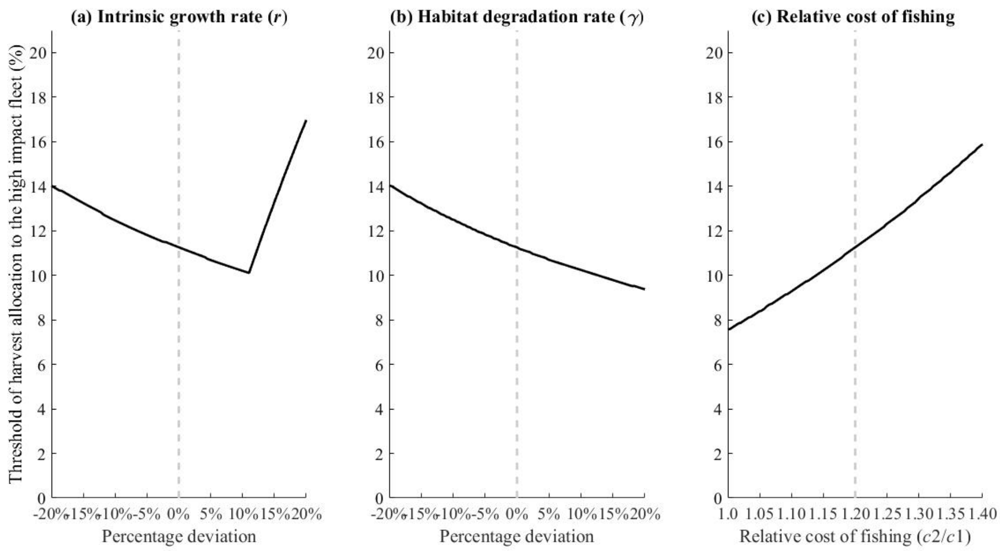
	Coefficient	Standard Error	
Intrinsic growth rate (*r*)	-0.042	0.006	***
Intrinsic growth rate squared (r^2)	0.0005	0.0004	
Habitat damage rate (γ)	-0.049	0.006	***
Habitat regeneration rate (ρ)	0.007	0.0003	***
Relative fishing cost (c^L/c^H)	0.410	0.020	***
Constant	-7.69	0.399	***
Log likelihood	-187.325		
Sample size	1000		

Note: This table reports the estimates of the coefficient and robust standard error from the generalised linear model with a logit link and the binominal family. All explanatory variables are measured in percentage, so that the coefficient represents a change in the proportion of catch allocated to the high impact fleet for one percentage change in the parameter value. *** 1% level, ** 5% level, and * 10% level.

3.4.5.3. Fishery collapse through suboptimal harvest allocation to the high impact fleet

We previously found in Section 4.3 that a suboptimal allocation of more than 10% to the high impact fleet in a slow-growing environment (ρ = 0.01) result in stock collapse. In this section, we examine the sensitivity of this result to ±20% deviations from the baseline values of the intrinsic growth rate (*r*) and the damage rate (γ) and the relative cost of fishing for the range $c^L/c^H \in [1,1.4]$. A relative cost of fishing equal to 1 means that the cost of fishing is the same for both fleets, while a relative cost of 1.4 means that the cost of fishing is 40% higher for the low impact fleet.

Figures 6: Sensitivity of the harvest allocation permitted to the high impact fleet before stock collapse occurs, in a slow-growing environment, ρ = 0.01, for ±20% deviations in the intrinsic growth rate (a), the habitat degradation rate (b), and for a range of relative fishing costs between the high and low impact fleets (c). The 0% marker in (a) and (b) and the relative fishing cost of 1.20 in (c) indicate the baseline value of these parameters.



For intrinsic growth rates lower than the baseline value, the level of allocation of harvest to the high impact fleet permitted before stock collapse occurs is higher than the baseline level (Figure 6a). This means that, for a slower-growing fish stock, a higher allocation of harvest to the high impact fleet is permitted than the baseline case before stock collapse occurs. This result reflects the fact that the fishery-wide optimal catch is set precautionary for the slow-growing stock, and so an allocation of harvest to the high impact fleet has relatively little impact on the stock. This result is consistent with the result in Figure 5 (a); for lower growth rates of the stock, the threshold of habitat regeneration rate below which a failure to account for habitat effects results in collapse of the fish stock is also lowered.

As the growth rate of the stock increases, this threshold of harvest allocation to the high impact fleet decreases (Figure 6a). However, when the growth rate is approximately 13% higher than the baseline, the relationship between the threshold and growth rate reverses; as the growth rate increases from this point, the permissible allocation of harvest to the high impact fleet before stock collapse occurs increases. This is because for sufficiently high stock growth rates, fish stock become resilient to the habitat degradation caused by fishing, and so a larger allocation to the high impact fleet is permitted. This again is consistent with the result in Figure 5 (a), where the threshold of habitat regeneration for the fast-growing stock was lowered. These results are also consistent with those seen in Table 2, where the relationship between the intrinsic growth rate and the optimal catch share to the high impact fleet was shown to be convex.

The threshold of harvest allocation to the high impact fleet is also sensitive to the damage rate of habitat as well as the relative cost of fishing between the high and low impact fleets. As the habitat damage rate increases, the permitted allocation to the high impact fleet decreases (Figure 6b). We also find that as the relative cost of fishing for the low impact fleet increases, the level of allocation to the high impact fleet that can be made without risking stock collapse will increase (Figure 6c). These results suggest that the stock is particularly susceptible to collapse due to the suboptimal allocation of harvest to the high impact fleet when the cost of fishing to the low impact fleet is relatively low, and where the damage rate is relatively high.

3.5. Discussion

Concern over the impacts of fishing on marine environments, coupled with a growing global awareness of the uncertain biological status of fisheries, has increasingly lead fishery managers to include the wider ecosystem impacts of fishing into their decision-making. In many jurisdictions, the need to also maximize the economic benefit from the fishery and to achieve social goals, such as maintaining employment or promoting the economic development of local communities, means that an understanding of how to manage habitat degradation in fisheries with multiple user groups is a high priority for fishery managers. In this paper, we develop new insights into the level of fishery-wide catch, and its allocation across different user groups, that maximizes the overall profit of the fishery where damaging interactions between fishing and habitat exist. We further analyze how fishery outcomes, as measured by equilibrium levels of biomass, carrying capacity and fishery-wide profit, are affected when actual catch shares deviate from profit-maximizing catch shares.

Our results highlight the importance of accounting for destructive impacts of fishing on habitat when assessing both the optimal total catch for the fishery and allocation of catch shares across different user groups. When damaging interactions between fishing and habitat exist, failure to account for these interactions in the determination of the total catch results in the depletion of fish biomass and decline in the overall profit of the fishery. These negative impacts will be particularly severe in environments where habitat is slow to recover and the damage rate of fishing is high. Ignoring habitat effects in such circumstances may result in fishery collapse. However, in a fishery where harvest is set to maximize economic profit, ignoring habitat effects may have less extreme consequences for fishery performance where the fish stock grows either relatively slowly (due to the precautionary assessment of harvest levels for slow growing stocks) or relatively quickly (due to the increased resilience to the habitat effect afforded to stocks with fast growth rates).

The optimal allocation of catch shares across different user groups in damage-prone environments is highly sensitive to inter-group differences in the impact of fishing on the environment and the cost of fishing, with changes in these technical and economic conditions having a relatively larger effect on optimal catch shares than either the intrinsic growth rate of the fish stock or the strength of habitat-fishery interactions. That said, when harvest is allocated to maximize the overall profit of the fishery, the optimal catch share of one of the user groups is frequently zero. In particular, it is likely to be optimal to make no allocation of harvest to the high impact, but more efficient, user group in environments where habitat is slow to recover from fishing-induced damage, the damage rate of fishing is high and the gain in efficiency associated with use of high impact technologies is small. An allocation of a positive catch share to this user group in such environments may result in a substantial decline in the biological outcomes (i.e., biomass and carrying capacity) and poor economic performance (i.e., fishery-wide catch and profit) for the fishery. By contrast, when the habitat is resilient to fishing-induced damage, it is likely that the entire catch is optimally harvested by the high impact, but more efficient, user group.

In many fisheries, however, the effective exclusion of one user group from the fishery may be unacceptable, reflecting either the need to maintain community fishing activity or avoid broader economic, social and political problems associated with shutting down parts of commercial fishing operations (Dichmont et al., 2010). This may result in the harvest being allocated in a manner inconsistent with the optimal allocation. The opportunity cost, in terms of foregone profit, and the biological risks of deviating from the optimal harvest allocation differ depending on the extent of habitat effects and whether the allocation is made to favor the high or low impact group. For example, a suboptimal allocation made to a high impact group in a vulnerable environment will increase the likelihood of fish stock collapse, although the extent of this will depend on the intrinsic growth rate, the habitat degradation rate and the relative fishing costs. Avoiding such allocations, based on knowledge of general indicators of the biological and economic conditions in a fishery, may offer a pragmatic and low cost way to reduce the risk of inadvertent stock collapse. This contrasts with the case of less vulnerable environments where comparable suboptimal harvest allocations to low impact, but less efficient, groups can improve the biological outcomes for the fishery with only small losses of fishery-wide profit.

Our results also have implications for the allocation of catch shares to secondary and new user groups. For example, the allocation of a share of catch to a community group in a fishery dominated by large-scale commercial users and where habitat is slow to recover from fishing-induced habitat damage has the potential to improve both economic performance and biological outcomes. This highlights the importance of community quota allocations as a policy mechanism not only for providing social and economic benefits to fishing communities but for also alleviating detrimental fishery-habitat interactions, the benefits of which can extend beyond the current needs of the given community (Gutiérrez et al., 2011). This win-win outcome, underpinned by the complementary relationship between community allocations and ecosystem improvement, is also found in the literature examining socio-ecological frameworks (Folke, 2007; Ostrom, 2009).

Overall, our results emphasize the importance of incorporating information on damaging interactions between fishing and habitat, as well as the economic characteristics of fishing by different user groups, in harvest allocation processes. Our results also strengthen the case for undertaking ecological risk assessments for fishing activity, in which the fishing impacts on ecological systems are assessed according to the risk that specific management outcomes are not achieved (Hobday et al., 2011). However, undertaking these assessments and including more information on a wider range of fishery characteristics in management procedures imposes additional costs on fishery management, both in terms of the additional information required and the potential increase in enforcement costs (Arnason, 1990; Dowling et al., 2013). Because of this, fishery managers often formulate harvest strategies based on proxy targets, or rely on simpler bioeconomic models that omit complexity (Pascoe et al., 2016; Punt et al., 2014; Rindorf et al., 2017). Knowing when more information needs to be included to underpin management decisions, and when it can safely be omitted, is therefore highly relevant to fishery managers who face a trade-off between costly information gathering and analysis and the risks associated with ecosystem damage and overfishing, and failure to realise the full benefits of fisheries (Little et al., 2016). Our results highlight the importance of research aimed at improving knowledge of the vulnerability to different types of fishing activity of different habitats, and suggest that priority should be given to better embedding key biophysical and economic relationships in bioeconomic models and other decision support tools in fisheries where stocks are reliant on vulnerable habitats, the damage rate of fishing is high and where there is little efficiency gain from the use of high impact technologies.

Several caveats need to be noted when our results are interpreted. First, our analysis focuses on habitat effects that manifest as changes in the carrying capacity of the population. However, the adverse impacts of fishing-induced habitat damage on the productivity of stocks may occur through different channels, such as changes in the intrinsic growth rate of the population as well as changes in the resilience of specific stocks to habitat degradation, and changes in the nature of interactions with other species (Airoldi et al., 2008; Dobson et al., 2006; Foley et al., 2012). Incorporating alternative forms of habitat effects is a potential extension of our model. Second, our analysis is confined to user groups that derive benefit from the fishery in the same manner. That is, we assume that the net benefit to each of the two user groups in this paper can be measured by the economic profit of fishing. This assumption may be acceptable for commercially-oriented users but not for other important user groups such as recreational and Indigenous fishers who derive cultural, health and livelihood benefits from fishing (Acott & Urquhart, 2014; Johnson, 2017). Further research is needed to examine the use of catch shares to achieve optimal fishery outcomes where fishing impacts habitat and where user groups derive value from fishing differently. Third, our fishery performance indicators are limited to biomass, carrying capacity and fishery-wide profits. Broadening this focus to encompass indicators that capture other aspects of performance of interest to managers and other stakeholders would help to ensure the relevance of our analysis. This paper revealed the trade-offs between achieving biological and economic outcomes when suboptimal allocations are made in fast-growing habitats and the potential for stock collapse in slow-growing environments. These results further suggest that fishery managers need to pay attention to a broader range of performance indicators, particularly for vulnerable and slow-growing habitats such as cold water coral reefs, where there is value in maintaining the environment independent of fishing activity (Armstrong et al., 2017). Fourth, it is evident that marine resources are increasingly shared by a growing number of potentially conflicting activities, including aquaculture, tourism, transport and energy sectors (Arbo & Thủy, 2016; Stepanova, 2015) and that these user groups benefit both directly and indirectly from habitat, and in turn impact habitat quality. Incorporating these interactions and feedbacks between wild fisheries and other sectors, and between these sectors and habitat, requires an extension of the conventional bioeconomic model that predominantly focuses on a single sector and ignores the cumulative pressures imposed by multiple sectors. Recent studies incorporating such multi-sector interactions include Sala et al., (2013) and Xuan and Armstrong (2017).

Appendix: Derivation of marginal equilibrium conditions

This appendix sets out the derivation of the marginal equilibrium conditions shown in Equations (7) and (8) of this essay. Given the optimisation problem specified in Equation (5), and the Lagrangian function specified in Equation (6), the derivation of the marginal equilibrium condition where there is no habitat effect (or where habitat effect exists but only the low impact gear is used) is as follows:

First Order Conditions in the steady-state:

$$\pi_{h^L} = \mu^x \tag{A.1}$$

$$\pi_x = \mu^x[(1+\delta) - (1+G_x)] \quad \backslash \tag{A.2}$$

Rearranging and simplifying A.2 to isolate μ^x yields:

$$\mu^x = \frac{\pi_x}{\delta - G_x} \tag{A.3}$$

Set A.3 equal to A.1:

$$\pi_{h^L} = \frac{\pi_x}{\delta - G_x}$$

$$\delta = \frac{\pi_x}{\pi_{h^L}} + G_x$$

The derivation of the marginal equilibrium condition where there is a habitat effect and a high impact gear is used:

First Order Conditions in the steady-state:

$$\pi_{h^H} = \mu^x + \mu^K D_{h^H} \tag{A.4}$$

$$\pi_x = \mu^x[(1-\delta) - (1+G_x)] \tag{A.5}$$

$$\mu^x G_K = \mu^K[(1+\delta) - (1+H_K - D_K)] \tag{A.6}$$

Rearrange A.4 to isolate μ^K:

$$\mu^K = \frac{\pi_{h^H} - \mu^x}{D_{h^H}} \tag{A.4A}$$

Rearrange and simplify A.6 to isolate μ^K:

$$\mu^K = \frac{\mu^x G_K}{\delta - H_K + D_K} \tag{A.6A}$$

Let A.4A equal A.6A and rearrange to isolate μ^x:

$$\frac{\pi_{h^H} - \mu^x}{D_{h^H}} = \frac{\mu^x G_K}{\delta - H_K + D_K}$$

$$\pi_{h^H} - \mu^x = \frac{\mu^x G_K D_{h^H}}{\delta - H_K + D_K}$$

$$(\pi_{h^H} - \mu^x)(\delta - H_K + D_K) = \mu^x G_K D_{h^H}$$

$$\pi_{h^H}(\delta - H_K + D_K) - \mu^x(\delta - H_K + D_K) = \mu^x G_K D_{h^H}$$

$$\pi_{h^H}(\delta - H_K + D_K) = \mu^x\left[G_K D_{h^H} + (\delta - H_K + D_K)\right]$$

$$\mu^x = \frac{\pi_{h^H}(\delta - H_K + D_K)}{G_K D_{h^H} + (\delta - H_K + D_K)} \tag{A.7}$$

Rearrange A.5 to isolate μ^x:

$$\mu^x = \frac{\pi_x}{\delta - G_x} \tag{A.8}$$

Set A.7 equal to A.8:

$$\frac{\pi_{h^H}(\delta - H_K + D_K)}{G_K D_{h^H} + (\delta - H_K + D_K)} = \frac{\pi_x}{\delta - G_x}$$

$$\frac{(\delta - H_K + D_K)}{G_K D_{h^H} + (\delta - H_K + D_K)} = \frac{\pi_x}{\pi_{h^H}(\delta - G_x)}$$

$$\frac{\pi_x}{\pi_{h^H}} = \frac{(\delta - G_x)(\delta - H_K + D_K)}{(\delta - H_K + D_K) + G_K D_{h^H}}$$

Chapter 4: How does a network of marine protected areas impact adjacent fisheries?

4.1. Introduction

Marine protected areas (MPAs) are increasingly established globally in an effort to improve habitat quality and biodiversity in the world's oceans (Lester et al., 2009; O'Leary et al., 2016). While the foremost reason for establishing an MPA is the achievement of biological and conservation outcomes (Agardy et al., 2011; Wells et al., 2016), MPAs are now not only expected to achieve these outcomes, but also generate benefits to various marine resource users, including fisheries (Watson et al., 2014). Initially, promotion of MPAs as a means to achieve improvements for both conservation and fishery outcomes predominantly focused on isolated no-use MPAs in which no extractive activity was permitted (Caveen et al., 2015; Halpern et al., 2010; Pauly et al., 2002). Although the conservation benefits that come from no-use MPAs are well-established in the literature, previous studies deliver mixed conclusions regarding the benefits generated to fisheries (Dunne et al., 2014; Hilborn et al., 2004; Hughes et al., 2016), such that the creation of non-use MPAs may lead to long-term conservation and fishery improvements, but the negative short-term consequences to fisheries may be considerable (Brown et al., 2015).[15]

The desire to minimise the trade-offs between the potentially conflicting goals of conservation improvements and fishery benefits has led to debate over how to design and implement protected areas which achieve conservation outcomes, while at the same time generate benefits for marine resource users (Agardy et al., 2011; Charles & Wilson, 2009; Gaines et al., 2010;

[15] These consequences may include fishing effort displacement (Horta e Costa et al., 2013), increased fishing costs (Hannesson, 1998) and short-term reduction in catch (Hilborn et al., 2006). See also the extensive bioeconomic literature (Grafton et al., 2005; Holland & Brazee, 1996; Smith & Wilen, 2003) exploring the impacts of no-use marine protected areas on fisheries.

Krueck. et al., 2017; Sala et al., 2002). The concept of MPA networks arose from this desire to avoid the trade-offs that may occur when implementing isolated, no-use MPAs. MPA networks are comprised of collections of individual MPAs capable of operating synergistically, at various spatial scales and with a range of protection levels designed to achieve objectives that a single MPA cannot (IUCN-WCPA, 2008). These networks have the potential to function collectively to facilitate both ecosystem and fishery improvements above those which might be expected from an isolated MPA (Ballantine, 2014; Gaines et al., 2003; Gaines et al., 2010; Horigue et al., 2015; Roberts et al., 2001; Roberts et al., 2018). Recognition that MPA networks may more effectively and efficiently achieve conservation and fishery objectives than isolated no-use MPAs do has been reflected in global agreements such as the Convention on Biological Diversity (CBD). In 2011, the CBD set the Aichi Biodiversity Targets, which aim for 10% of coastal and marine areas to be placed within networks of protected areas by 2020 (CBD, 2011). The target also expressly includes the need to reconcile conservation of the environment with maintaining the benefits from ecosystem services (Spalding et al., 2013), of which fisheries comprise a major part (Caveen et al., 2015). In response, there has been a global increase in the number of MPA networks, with countries including New Zealand, Australia and the United Kingdom implementing large-scale MPA networks within their jurisdictions.

There is a commensurate increase in the amount of research discussing the design of MPA networks to achieve improved conservation and fishery outcomes (Almany et al., 2009; Rassweiler et al., 2014; Rassweiler et al., 2012; Roberts et al., 2018; Smith & Metaxas, 2018). However, the empirical literature evaluating the ex post impacts of an MPA network comprised of MPAs of varying levels of protection on fisheries are relatively scarce. Previous studies examining the effects of MPA networks for fisheries confine their examination to: networks of no-use MPAs (Gell & Roberts, 2003; Hopf et al., 2016b); a specific species (Harrison et al., 2012; Williams et al., 2009); fisher responses and perceptions regarding MPA networks (Arias et al., 2015; Ayer et al., 2018; Cabral et al., 2017; Ordoñez-Gauger et al., 2018); or ex ante evaluation of the potential long and short-term consequences of a network for fisheries (White et al., 2013). One exception is Reimer and Haynie (2018) who provided an ex post evaluation of the economic impacts on adjacent fisheries of MPAs, the design of which consists of various protective measures in the large area of the North Pacific Ocean for the conservation of Steller sea lions.

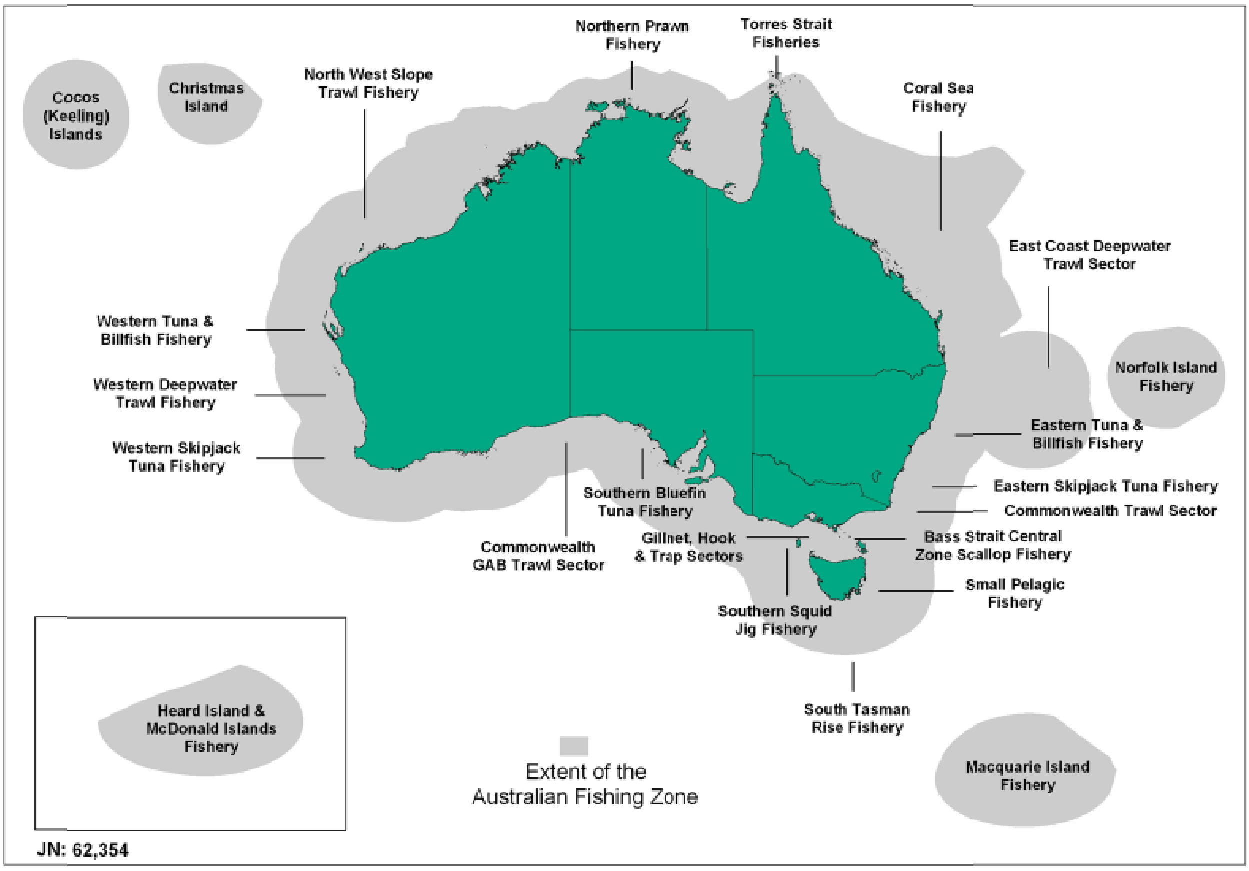
The aim of this paper is to extend the above literature by assessing the effect of an MPA network consisting of both no-use and multi-use MPAs on adjacent fisheries, taking Australia's south-east marine reserve network (SEMRN) as a case study. This network was established in 2007 in the Commonwealth-managed waters around Tasmania, South Australia, Victoria and New South Wales, in waters shared with a number of Commonwealth-managed fisheries, including the highly valuable Southern and Eastern Scalefish and Shark Fishery (SESSF) and the Southern Bluefin Tuna Fishery (SBTF). The network was primarily established for habitat conservation, although it also formed part of a package of management changes in Commonwealth-managed fisheries. The management changes occurring in the same time period indicate an increase in the level of precaution used when managing Commonwealth fish stocks, which may also be expected to impact fishery performance (Punt, 2006). We use a panel of data comprised of eight fisheries and thirty-nine species with a time series of fourteen years (2001-2015) and apply a difference-in-differences (DiD) modelling approach to isolate the effect this network had on the performance of adjacent fisheries. Given this network's geographical location is such that there exists a treatment and control group of fisheries, the SEMRN provides an appropriate and useful case study for exploring potential fishery impacts of an MPA network. We take as our performance indicators the catch and gross value of production (GVP) of these fisheries. These metrics, often driven by key commercial species within fisheries, are a common measure of fishery performance in Australian fisheries (Flood et al., 2016) and so provide a convenient measure of fishery performance in this context. We use this dataset and methodology to answer three research questions; first, has there been any effect, as assessed by a change in either catch or GVP, for the treatment group of species resulting from the SEMRN? Second, given the change to fishery management occurring in the same time frame may be expected to impact the chosen performance indicators, does the effect from the SEMRN persist once other management changes are accounted for? Third, has the network had any impact specific on the key commercial species responsible for driving fishery performance, i.e. is there any indication that highly valued species have been more or less affected by the establishment of the SEMRN, relative to other species in the sample?

4.2. Background: Australian context

The Australian federal government (the ‘Commonwealth’ government) is responsible for managing the marine resources within the boundaries of Australia’s exclusive economic zone (EEZ), excluding those resources which lie within twelve nautical miles of the coastline which are managed by the states and territories. The Commonwealth-managed fisheries have been the responsibility of the Australian Fisheries Management Authority (AFMA) since 1991 (Figure 1).

Figure 1: Commonwealth-managed fisheries

(Source: AFMA http://www.agriculture.gov.au/fisheries/domestic/zone)



In late 2005, due to concerns over unsustainability and unprofitability in Commonwealth-managed fisheries, three major initiatives were announced: (1) a statutory direction from the Minister of Fisheries to the AFMA to recover overfished stocks and prevent future overfishing (the introduction of the harvest strategy policy being a core aspect of this direction), (2) a structural adjustment package to remove excess effort from specific fisheries, and (3) the introduction of an MPA network in south-east Australia (Rayns, 2007). While the harvest strategy policy and structural adjustment package were intended as fisheries management reform, which is the responsibility of the AFMA, the MPA network was established and administered under the Environment Protection and Biodiversity Conservation (EPBC) Act 1999 and is the responsibility of the Director of National Parks. The rest of this section discusses first the implementation and features of this MPA network, known as the south-east marine reserve network, and then the introduction of the harvest strategy policy and structural adjustment package.

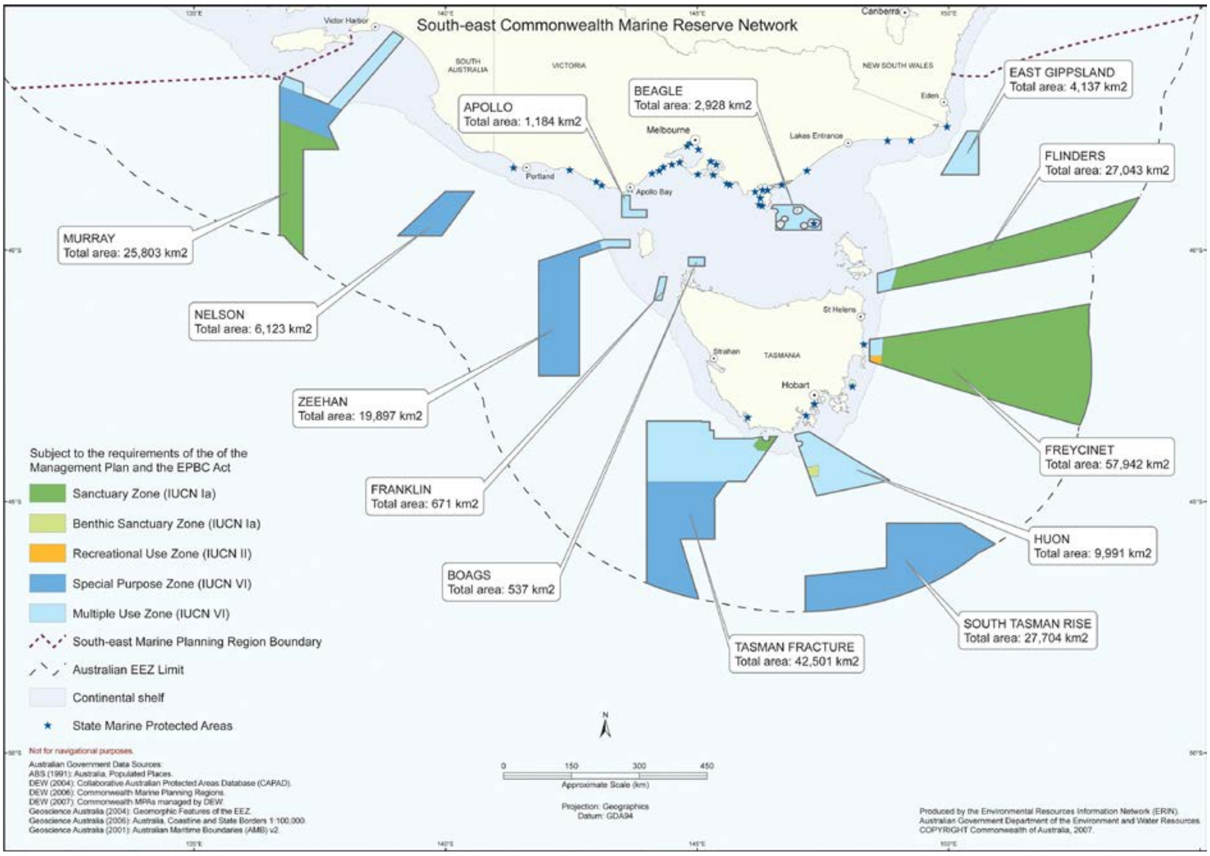
4.2.1. The South-East Marine Reserve Network (SEMRN)

The SEMRN, as originally proposed by the Australian government in December 2005, was to cover approximately 170,000 square kilometres of ocean in the south-east region off the coasts of South Australia, Tasmania, Victoria and New South Wales. This network was the first step in creating the National Representative System of Marine Protected Areas (NRSMPA) in the Commonwealth marine jurisdiction, which the Australian government had committed to establishing by 2012 as part of their commitment to the CBD. The goal of the NRSMPA has been to contribute to the conservation and maintenance of marine ecosystems and biodiversity in the long-term, while minimising any adverse impacts on marine users, both commercial and recreational (ANZECC-TFMPA, 1998). After the SEMRN was announced, a study was commissioned to investigate the anticipated impact of the SEMRN to adjacent fisheries and the socio-economic consequences to fishing communities (Buxton et al., 2006). This study found that the combined effect of the MPAs had potentially high socio-economic consequences for rural fishing communities, due to effort displacement in the highly valuable blue-eye trevalla, blue grenadier, scallop and tiger flathead fisheries, among others. The report also found that the SEMRN could be redesigned to avoid these impacts, without compromising the conservation goals of the NRSMPA. After boundary changes and re-zoning of certain MPAs, the SEMRN was proclaimed in June 2007. This re-designed network consists of fourteen MPAs, covering approximately 388,464 square kilometres, or 23.7% of the south-east marine region (Figure 2).[16]

[16] Not shown in Figure 2 is the Macquarie Island Commonwealth Marine Reserve, which was established in October 1999.

Figure 2: Australia's south-east marine reserve network

(Source: Proclamation of Apollo Commonwealth Marine Reserve, Explanatory Statement https://www.legislation.gov.au/Details/F2007L01910/Download)



Each of the fourteen MPAs is assigned to an IUCN category, according to its management objectives. The strictest IUCN category, Ia, permits no commercial-scale fishing, while the least stringent category, VI, permits a wide range of activities within the MPA area so long as those activities do not significantly damage the seafloor or other values of the area. Of the total area under protection, 12% is designated multiple use, 48% is designated for either habitat protection, recreation or special purpose, and 40% declared either a sanctuary or a national park (Parks, 2013).[17] Although the management plan for this network did not come into effect until 2013, management of these MPAs in the interim was in accordance with their IUCN category

[17] IUCN categories not represented in the SEMRN are categories Ib (wilderness area), III (natural monument or feature) and V (protected landscape or seascape) (Day et al., 2012).

under the EBPC Act.[18] The management plan made no changes to the design or categorisation of the MPAs, but formalised the monitoring and enforcement of the network.

4.2.2. Fisheries management changes

In November 2005, the Australian government announced the Securing Our Fishing Future initiatives, aimed at reducing the level of fishing effort and the development of a formal harvest strategy policy for Commonwealth fisheries. A key part of this package was a structural adjustment package, to allow fishers to voluntarily exit targeted fisheries and so reduce the overall level of fishing effort through a vessel buyback scheme. The fisheries targeted in this scheme were the Bass Strait Central Zone Scallop Fishery (BSCZSF), the Eastern Tuna and Billfish Fishery (ETBF), the Northern Prawn Fishery (NPF) and the trawl, gillhook and trap sectors of the SESSF.[19] The scheme ultimately resulted in a 35% reduction in fishing concessions for the SESSF, a 40% reduction for the NPF, a 47% reduction for the ETBF and a 14% reduction for the BSCZSF (Vieira, 2010). The other major change to Australian fishery management was the development of the Commonwealth Harvest Strategy Policy in 2007. This policy formalised the framework for developing harvest control rules for all commercially harvested species in Commonwealth fisheries, and came into effect September 2007, with full implementation required by January 2008. Prior to this policy, there was no guidance as to how to achieve economic efficiency or ecologically sustainable development for Commonwealth fisheries (Rayns, 2007). A formal harvest strategy was applied in the SESSF in the 2005 fishing season on the advice of the SESSF resource assessment group. The SESSF is a quota-managed fishery, and so requires a total allowable catch (TAC) to be set for a species; prior to 2005, the determination of this TAC was informal and adhoc, resulting in overfishing (Smith et al., 2014;

[18] See *Environment Protection and Biodiversity Conservation Regulations 2000,* Schedule 8.
[19] These fisheries were targeted due to concerns of overfishing, while fishers in other Commonwealth fisheries were also eligible to enter the scheme if they wished.

Smith et al., 2008). The 2005 harvest strategy adopted a 'tiered' approach, where the harvest control rule determining the TAC is more or less rigorous depending on how much information is available for assessing stock status. The harvest strategy also set lower limits and targets for biomass, with the target biomass being the maximum economic yield (MEY) for a species.[20]

4.3. Methods

The difference-in-difference (DiD) approach finds the impact of a policy change by analysing differences in a treatment group, where the treatment group is expected to have been affected by the policy, before and after the policy change, along with differences in a control group at matching times.[21] We take catch and gross value of production (GVP) as the dependent variables in the DiD model to assess the potential impact of the South-east marine reserve network for the treatment group, which corresponds to an econometric test of whether the policy treatment causes a change in the catch or GVP relative to the change in the control group within the same time frame. We use this analysis to explore changes for the treatment group, with and without controlling the confounding effect of changes in fisheries management, and then extend the model to explore heterogeneity in the treatment effect for commercially valued species.

4.3.1. Average treatment effect

To find the average treatment effect for the treatment group, we estimate a pure DiD model shown by equation (1) below:

$$Y_{i,s,t} = \alpha + \beta_1 POST2007_t + \beta_2 TREAT_{i,s} + \beta_3 POST2007_t \times TREAT_{i,s} + \theta_t + \sigma_i + \delta_s + \varepsilon_{i,s,t} \tag{1}$$

[20] Where information regarding the calculation of MEY is not available, the maximum sustainable yield (MSY) is used as a proxy; here, MEY is equal to 1.2MSY.
[21] The DiD approach has diverse application in economics, with the approach used to analyse policy changes in areas such as education (Card & Krueger, 1992) and health economics (Wing et al., 2018).

$Y_{i,s,t}$ is the catch or GVP for fishery *i*, species *s* and year *t*. $POST2007_t$ and $TREAT_{i,s}$ are dummy variables equal to one indicating an observation occurs after the SEMRN was implemented and that it is a treatment fishery, respectively. $POST2007_t \times TREAT_{i,s}$ is an interaction variable, the coefficient of which gives the average treatment effect of the SEMRN. Year fixed effects (θ_t), fishery fixed effects (σ_i) and species fixed effects (δ_s) control for all time-invariant factors that affect both treatment and control groups. Factors controlled for by these fixed effects include time trends related to unobserved factors (year fixed effects), differences in efficiency of fishing gear used or in scale between fisheries (fishery fixed effects)[22] and physiological differences between species which remain constant over time (species fixed effects). Table 1 shows the interpretation of the coefficients in the DiD model and the derivation of the DiD estimator.[23]

Table 1. Interpretation of coefficients in equation (1) and derivation of DiD estimator

	Pre-SEMRN	Post-SEMRN	Difference over time
Control group	α	$\alpha+\beta_1$	β_1
Treatment group	$\alpha+\beta_2$	$\alpha+\beta_1+\beta_2+\beta_3$	$\beta_1+\beta_3$
Difference across groups	β_2	$\beta_2+\beta_3$	β_3

From Table 1, we see that α gives the measure of the dependent variable, averaged over species, fisheries and time, for the control group, in the pre-SEMRN time period. $\alpha+\beta_2$ gives the measure of the dependent variable for the treatment group in the pre-SEMRN period. The

[22] The fisheries in this sample (detailed further in section 4) differ significantly in their characteristics, with some fisheries targeting one species and others targeting multiple species. The fisheries also vary in their gear types, input and output controls and length of fishing seasons. These differences are also controlled by fishery fixed effects.

[23] Adapted from Wooldridge, (2013), p457.

difference between the control and treatment groups in the pre-SEMRN period is therefore given by β_2. The measure of the dependent variable for the control group in the post- SEMRN period is given by $\alpha + \beta_1$, with β_1 showing the change that will occur independent of the SEMRN. The measure of the dependent variable for the treatment group in the post- SEMRN period is given by $\alpha + \beta_1 + \beta_2 + \beta_3$, with $\beta_1 + \beta_3$ showing the change that will occur in the treatment group over time. The difference between the control and treatment groups in the post-SEMRN period is given by $\beta_2 + \beta_3$. Therefore, the difference between groups attributable to SEMRN, i.e. the DiD estimator, for equation (1) is given by β_3, and is interpreted as showing the average treatment effect for each species in the treatment group, for every year in the post-SEMRN period. The null hypothesis for equation (1) is that the DiD estimator is equal to zero, and so the SEMRN has had no impact on the treatment group.

4.3.2. Controlling for fishery management changes

To explore the potential confounding effect of fishery management changes during the same time series, we estimate a modified DiD model with control variables included for the introduction of the harvest strategy policy and the vessel buyback scheme. These management variables are included in addition to the year, fishery and species fixed effects as these do not control for aspects of fishery management that have changed over time or management changes that affect a subgroup of fisheries in the sample. The modified DiD model is given as:

$$Y_{i,s,t} = \alpha + \beta_1 POST2007_t + \beta_2 TREAT_{i,s} + \beta_3 POST2007_t \times TREAT_{i,s} + z'_{i,s,t}\gamma + \theta_t + \sigma_i + \delta_s + \varepsilon_{i,s,t} \qquad (2)$$

where $z_{i,s,t}$ is a 2 × 1 vector of explanatory variables indicating fishery management changes that occurred during the time series. The first is a dummy variable indicating the application of the harvest strategy policy for a given species and fishery in a given year. The second is a dummy variable equal to one if a fishery in question was part of the vessel buyback scheme. The DiD estimator for equation (2) is also given by β_3, with a null hypothesis that the DiD estimator is equal to zero, and so the null hypothesis for equation (1), that there has been no effect due to the SEMRN, is unchanged by controlling for management changes.

4.3.3. Heterogeneous effects for commercially valued and less-valued species

To explore the potential for a difference between commercially valued and less-valued species in the treatment group (i.e. 'heterogeneous treatment effects'), we use a modified DiD model as shown in equation (3) below:

$$\begin{aligned} Y_{i,s,t} = \alpha + \beta_1 POST2007_t + \beta_2 TREAT_{i,s} + \beta_3 POST2007_t \times TREAT_{i,s} \\ + \beta_4 CV_{i,s} + \beta_5 POST2007_t \times TREAT_{i,s} \times CV_{i,s} + \theta_t + \sigma_i + \delta_s + \varepsilon_{i,s,t} \end{aligned} \quad (3)$$

where $CV_{i,s}$ is a dummy variable equal to one if a treatment species is considered commercially valued. The interpretation of the DiD estimator, β_3, remains the same as in equations (1) and (2). The coefficient of the commercially valued interaction variable, β_5, gives the additional treatment effect for the species deemed commercially valued, and thus shows the heterogeneity between the average treatment effect for all species in the treatment group, and the average treatment effect for those commercially valued species in the treatment group. We test two hypotheses. First, we test a null hypothesis that the DiD estimator for this model, given by β_3, is zero, and so the null hypothesis for equations (1) and (2), that there has been no effect due to the SEMRN, is unchanged after accounting for possible heterogeneity in treatment effects within the treatment group. The second hypothesis is that the coefficient on the CV interaction variable, β_5, is equal to zero, such that there is no heterogeneity in treatment effects.

4.4. Data

Data on the catch (in tonnes) and GVP (in nominal AUD) for Commonwealth fisheries is drawn from the Australian fisheries statistics reports for 2012 (Skirtun et al., 2013) and 2015 (Savage, 2015). These reports provide data for eight Commonwealth fisheries and thirty nine species, for the financial years 2001/02 to 2014/15.[24] To assign fisheries and species into treatment and control groups, first the boundaries of the SEMRN were found using the Collaborative Australian Protected Areas Database (CAPAD) on MPAs for 2008 (CAPAD, 2008). Then, the location of fishing activity for each fishery was determined using the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) fishery status reports for 2006 (Larcombe & McLoughlin, 2007) and 2008 (Wilson et al., 2009).[25] Fisheries were assigned into either the treatment or control groups by comparing the location of fishing effort in each fishery as shown in the ABARES reports, with the boundaries of the MPAs within the SEMRN shown in Figure 2. The data required to compare the distance between fishery boundaries and the SEMRN was not available and so fisheries adjacent to the boundaries of the SEMRN were placed in the treatment group, while those further away were placed in the control group, with individual species assigned to a group depending on which fishery they were caught in. Given the spatial scale of the fisheries and SEMRN, the consistency in fishery boundaries and the lack of large-scale fishing effort migration over time, this method was deemed appropriate for the purposes of this analysis.[26]

[24] This does not include fisheries and species which are grouped together for the report, due to lack of specificity. We also exclude the Bass Strait Central Zone Scallop Fishery, as this fishery was subject to various closures independent of the SEMRN throughout the time series. GVP data is unavailable for the Western Tuna and Billfish Fishery from the 2008/09 financial year onwards, and catch and GVP data is unavailable for the Southern Squid Jig Fishery from 2012/23 onwards, due to confidentiality requirements.

[25] The ABARES fishery status reports between the years 2000/01 and 2016 were also examined to ensure the choice of treatment and control fisheries remained appropriate for the entire sample period.

[26] A potential extension of this paper may be to make a more robust allocation of fisheries into treatment and control groups, perhaps using effort location data.

Fishing effort location migrated over the time series, but the extent of migration is not substantial such that treatment group fisheries became control group, and vice versa.

The time series is split into pre- and post-SEMRN periods at the 2008/09 financial year, reflecting the fact that, while the SEMRN was proclaimed in June 2007, it did not commence until August 2007.[27] As such, the 2008/09 financial year is the first in the time series entirely subsequent to the SEMRN proclamation. Sensitivity of results to both this choice of post-SEMRN period and to the definition of the control and treatment groups is given in Section 4.5.4.

Table 2. Management characteristics of fisheries in sample, by group

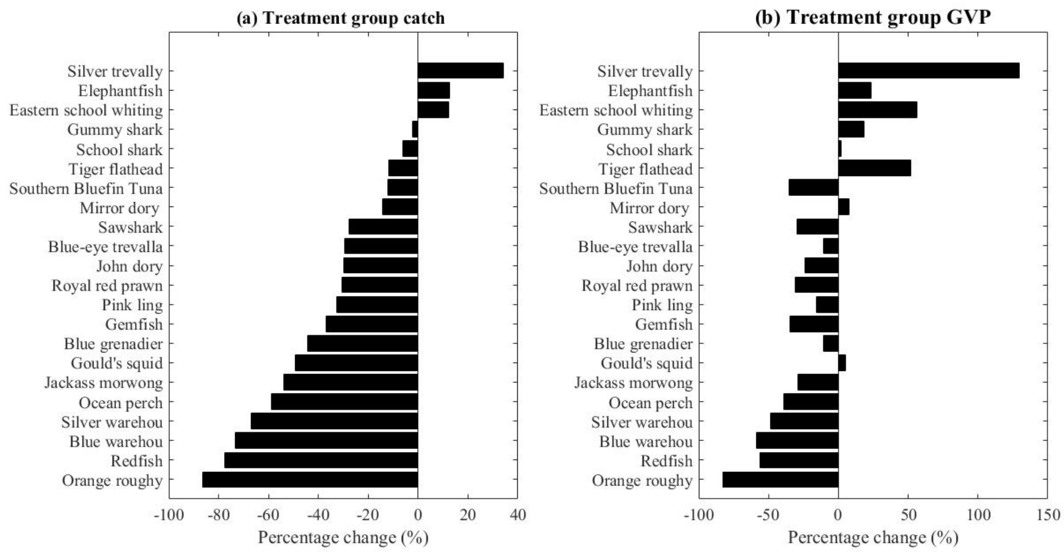
(a) Treatment fisheries	Vessel buyback scheme	Year harvest strategy policy implemented†	Gear type
Southern and Eastern Scalefish and Shark Fishery (Commonwealth Trawl sector)	Yes	2005	Bottom trawl, Danish seine
Southern and Eastern Scalefish and Shark Fishery (Gillhook and Trap sector)	Yes	2005	Gillnet, Longline, Trap
Southern Bluefin Tuna Fishery	No	n/a^	Pelagic longline, Purse seine
Southern Squid Jig Fishery	No	2007	Jig
(b) Control fisheries			
Southern and Eastern Scalefish and Shark Fishery (Great Australian Bight sector)	No	2005	Bottom trawl, Danish seine
Northern Prawn Fishery	Yes	2007	Otter trawl
Western Tuna and Billfish Fishery	No	n/a^^	Pelagic longline
Eastern Tuna and Billfish Fishery	Yes	2010	Pelagic longline

† Source: Ward et al. (2013)
^ Commission for the Conservation of Southern Bluefin Tuna (CCSBT) harvest strategy implemented in 2011/12 season
^^ Harvest strategy was developed, but has not been implemented (http://www.afma.gov.au/sustainability-environment/harvest-strategies/)

[27] Proclamation of Apollo Marine Park, https://www.legislation.gov.au/Details/F2017C00985, retrieved 19 July 2018.

A summary of the management and effort features of each fishery is given in Table 2. Of the eight fisheries in the sample, four fisheries were part of the vessel buyback scheme, two each from the treatment and control groups. In addition to this, six of the fisheries in the sample have been subject to the harvest strategy policy at different stages in the time series.

Figures 3: Percentage change in average catch (a) and GVP (b) for species in treatment group.



Figures 3 shows the percentage change in average catch and GVP for species in the treatment group. Of the twenty-two species in the treatment group, only three species (eastern school whiting, silver trevally and elephantfish) experienced an increase in average catch in the post-SEMRN period (Fig. 3a). Certain species also experienced more severe declines in catch than others; blue grenadier, silver warehou, orange roughy, jackass morwong, redfish, ocean perch, blue warehou and Gould's squid all suffered a greater than 40% decline in catch in the post-SEMRN period. A similar pattern is found for GVP, although not as extreme (Fig. 3b). While the majority of species also experienced a decline in GVP in the post-SEMRN period, eight species experienced an increase. For example, the GVP of silver trevally increased by 130%, and tiger flathead and eastern school whiting increased in value by 50%

Figures 4: Percentage change in average catch (a) and GVP (b) for species in control group.

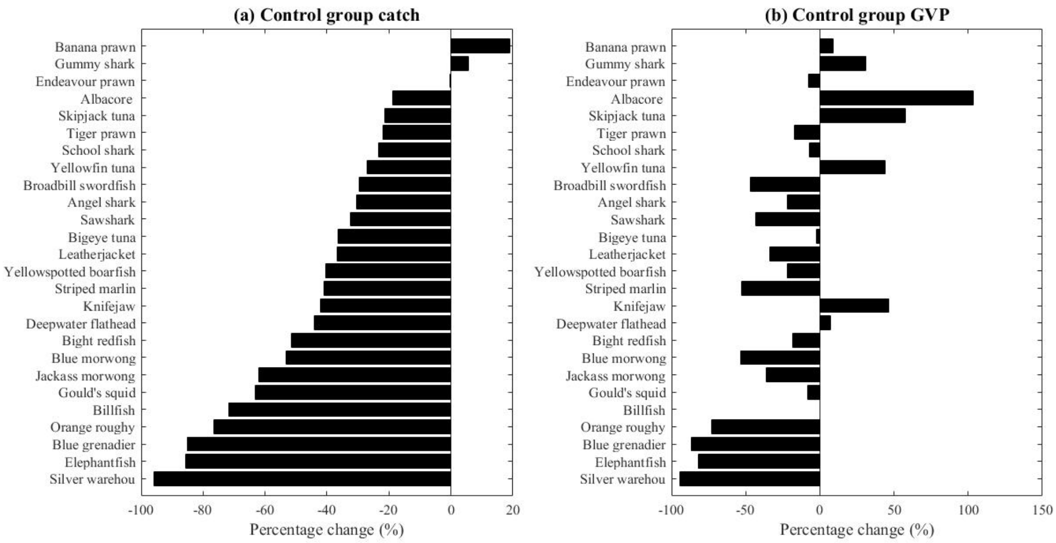


Figure 4 shows the percentage change in average catch and GVP for species in the control group. Similar to the treatment group, while some species experienced increases in catch in the post-SEMRN period (i.e. banana prawn and gummy shark), the majority of species in the group declined in catch, with silver warehou, elephantfish and blue grenadier experiencing a more than 80% decline in catch (Fig.4a). Again, there is similar but less extreme pattern found for GVP; where only two species experienced increases in catch in the post-SEMRN period, seven species experienced increases in GVP after the SEMRN was established, including albacore and skipjack tuna (103% and 57%, respectively) and deepwater flathead (7%).

To identify the commercially valued treatment species in the sample, the proportion of value contributed to the GVP of each fishery by each species, in the 2006/07 financial year (the year before the SEMRN was proclaimed), was examined and those species contributing over 20% to the GVP of the fishery in which they are targeted were selected. The commercially valued species in the treatment group are blue grenadier, tiger flathead, gummy shark, blue-eye trevalla and southern bluefin tuna.[28] We also observe a relatively higher decline in performance measures for commercially valued species in the post-SEMRN period than other species, suggesting the possibility of heterogeneity in treatment effects within the treatment group. The average catch and GVP of commercially valued species declined by 786 tonnes and $3 million, respectively; compared to a decline in the treatment group overall of 451.8 tonnes and $1.18 million.

4.5. Results

4.5.1. Average effect of the SEMRN on adjacent fisheries

Table 3 shows the DiD regression results for equation (1).[29] The results show that there is a statistically significant negative treatment effect for catch, suggesting that the SEMRN has had a negative impact on catch for adjacent fisheries. More specifically, catch of the treatment species was 245 tonnes lower per year on average than catch of the control species after the SEMRN was established. While we see a statistically significant negative treatment effect on catch, we do not see a significant effect for GVP. The DiD estimator for GVP suggests that in the sample, the SEMRN had a negative impact of $323,000 on the gross value of production on average, but that this negative treatment effect is statistically insignificant.

[28] Blue grenadier, tiger flathead and blue-eye trevalla were among those species identified by Buxton et al., (2006) as valuable species when the SEMRN was redesigned; see section 2.1.
[29] All regression results presented in section 4.5 have time, fishery and species fixed effects included.

Table 3. DiD estimation results for equation (1)

	Catch (tonnes)	GVP ($AUD000's)
Post-2007	-341.93** (159.52)	-2917.3** (1203.9)
Treatment	-810.88*** (157.21)	-29807.6*** (2037)
Post-2007×Treatment (DiD Estimator)	-245.78** (98.45)	-323.8 (616.1)
Constant	1853.24*** (166.34)	33658.8*** (2297)
R-squared	0.71	0.84
Number of pooled observations	813	782

Note: Robust standard errors in parentheses. * significant at 10% level, **significant at 5% level, and *** significant at 1% level.

Table 4. DiD estimation results when controlling for fishery management changes

	Catch (tonnes)			GVP ($AUD000's)		
Post-2007	-341.93** (159.52)	-344.00** (161)	-344.00** (161)	-2917.28** (1203.89)	-6136.27*** (1594.14)	-6136.27*** (1594.14)
Treatment	225.03 (188.56)	-810.73*** (157.18)	223.51 (205.09)	-6549.51*** (2531.15)	-29748.24*** (2159.12)	-7843.15*** (2257)
Post-2007×Treatment (DiD Estimator)	-245.78** (98.45)	-246.02** (97.29)	-246.02** (97.29)	-323.78 (616.12)	-358.36 (611.42)	-358.36 (611.42)
Vessel buyback fishery	1035.91*** (168.9)		1034.24*** (187.25)	23258.10*** (3186.41)		21905.09*** (3047.64)
Harvest Strategy Policy		2.59 (86.97)	2.59 (86.97)		3635.42*** (1136.37)	3635.42*** (1136.37)
Constant	817.33*** (203.56)	1852.95*** (168.26)	818.71*** (209.65)	10400.70*** (2718.01)	33413.99*** (2377.48)	11508.90*** (2433.05)
R-squared	0.71	0.71	0.71	0.84	0.84	0.84
Number of pooled observations	813	813	813	782	782	782

Note: Robust standard errors in parentheses. * significant at 10% level, ** significant at 5% level, and *** significant at 1% level.

4.5.2. Confounding effects of fisheries management changes

Table 4 presents the estimation results of equation (2) which includes additional explanatory variables representing the vessel buyback scheme and the harvest strategy policy. In this model specification, we control for the possibility that the negative coefficient of the DiD estimator found in the previous section is driven by two major fishery management changes that occurred during the same time frame as the SEMRN. The result shows that the inclusion of these additional control variables does not influence our result that the SEMRN has had a significant negative effect on catch, but not GVP, for the treatment group. The sign and statistical significance of the DiD estimator is robust to the additional explanatory variables across all regressions. Moreover, the size of the estimated treatment effect is not affected by the additional variables, which suggests that while these additional variables are statistically significant, they do not increase the explanatory power of the model to a substantial degree, due to the collinearity between new and old variables.[30]

The coefficient on the vessel scheme variable are positive and highly significant ($p<0.01$), for both catch and GVP. The estimated coefficient on this variable suggests that the species caught in fisheries involved in the vessel buyback scheme had, on average, a catch 1035 tonnes and a GVP $23 million higher than species caught in fisheries who were not part of the scheme. The coefficients on the harvest strategy variable are positive for both dependent variables, although is only statistically significant for GVP. The coefficient on the harvest strategy variable in the

[30] The coefficient for the treatment variable signifying whether a species belongs to the treatment group of fisheries, shows some variability when the vessel buyback variable is included in the analysis. This may be because the vessel buyback variable controls for four fisheries which were part of the scheme, two of which are in the treatment group. The collinearity between this variable and the treatment variable (r = 0.65) may be leading to this change in the coefficient. In addition, the fishery fixed effects included in each regression have different coefficients across equations (1) and (2) where the vessel buyback variable is included, which may also lead to differences in the treatment coefficient.

GVP regressions suggests that the species under the harvest strategy policy has, on average, a GVP \$3.6 million higher than species which were not subject to the policy.

4.5.3. Heterogeneity in treatment effect

In this section, we explore the potential for heterogeneous treatment effects between the commercially valued species in the treatment group and other species within that same group. The results presented in Table 5 give the estimation results for equation (3). The coefficient on the dummy variable indicating a commercially valued species (*CV*) is positive and highly significant ($p<0.01$) for both catch and GVP, suggesting that catch and GVP for commercially valued species were 2365 tonnes and \$8.2 million higher on average than less-valued species. We also confirm the result from Sections 5.1 and 5.2 above, where the SEMRN had a negative effect on catch for the treatment group, but no significant effect on GVP. The DiD estimate for catch retains the statistical significance from equation (1), although the magnitude of the estimate has decreased slightly. Further, the coefficient on the CV interaction variable is statistically insignificant for both catch and GVP, suggesting there exists no heterogeneity in treatment effects, that is, commercially valued species were not impacted by the SEMRN more or less than the treatment group of species overall.

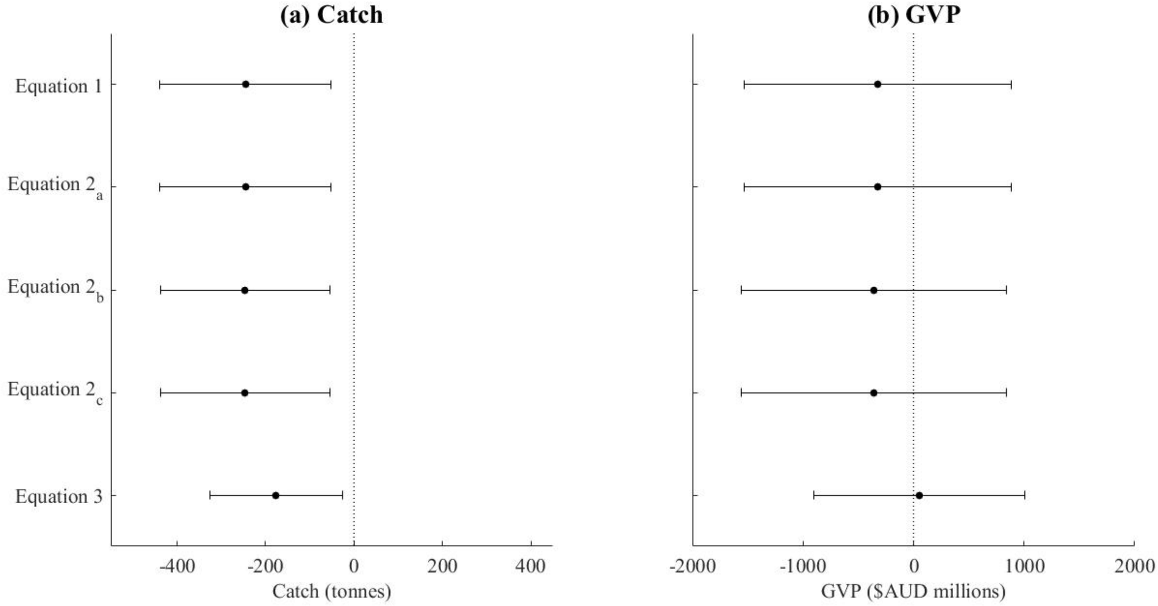
Table 5. DiD estimation results for heterogeneous treatment effects

	Catch (tonnes)	GVP ($AUD000's)
Post-2007	-335.01** (146.89)	-2891.16** (1186.74)
Treatment	-834.65*** (157.03)	-29937.98*** (2030.85)
CV	2365.58*** (466.82)	8243.76*** (1229.93)
Post-2007×Treatment (DiD Estimator)	-176.25** (76.11)	51.1 (486.49)
Post-2007×Treatment×CV (CV interaction)	-303.8 (244.84)	-1612.66 (1515.23)
Constant	1851.88*** (157.28)	33654.51*** (2294.86)
R-squared	0.76	0.85
Number of pooled observations	813	782

Note: Robust standard errors in parentheses. * significant at 10% level, ** significant at 5% level, and *** significant at 1% level.

Figures 5 (a,b) below summarise the results presented in Tables 3, 4, and 5. These figures show the coefficient of each DiD estimate with 95% confidence intervals across equations (1), (2) and (3). These figures demonstrate the relatively low level of variability across DiD estimates, indicating the estimated impact of the SEMRN on catch and GVP is robust to both the inclusion of variables controlling for the introduction of the harvest strategy policy and vessel buyback scheme in the same timeframe, and after controlling for possible heterogeneity in treatment effects.

Figures 5: DiD estimator with 95% confidence intervals for equations (1), (2) and (3) for catch (a) and GVP (b) dependent variables. The subscript a, b and c on equation (2) refers to equation (2) with vessel buyback variable, harvest strategy policy variable, and both variables respectively. GVP is scaled to $AUD millions for clarity.



4.5.4. Sensitivity analysis

Finally, we examine the sensitivity of our results to the choice of treatment group and post-SEMRN year. The estimation of treatment effects may be influenced if the treatment group includes species whose catch was lower due to constraints on the total allowable catch independent of the SEMRN, if fisheries which ought not to be in the treatment group are included, and if the post-SEMRN time period is incorrectly specified. To address these issues, we re-estimate equation (1) where (i) those treatment species under rebuilding plans during the sample period (orange roughy, school shark, sawshark and gemfish) are removed from the sample, (ii) the treatment fisheries are limited to the CTS and GHTS sectors of the SESSF (i.e. the other treatment fisheries, SBTF and SSJF are removed from the sample), and (iii) the post-

SEMRN year is the 2007/08 financial year. The DiD estimators for the baseline and re-estimated models are shown in Table 6.

Table 6. Sensitivity of DiD estimates

	Catch (tonnes)	GVP ($AUD000's)
Baseline (Table 5)	-245.78**	-323.78
	(98.45)	(616.1)
Rebuild species removed	-234.09**	-224.34
	(107.38)	(676.76)
SESSF only	-232.79**	375.03
	(101.90)	(537.9)
Post-SEMRN year	-284.58***	-198.42
	(105.61)	(630.19)

Note: Robust standard errors in parentheses. * significant at 10% level, ** significant at 5% level, and *** significant at 1% level.

We find overall that the baseline result, that there is a significantly negative treatment effect for catch, but not for GVP, as a result of the SEMRN, is robust to these re-estimations. When rebuilding species are removed from the sample, we see similar treatment effects for both catch and GVP, as compared to the baseline model. As one would expect, the DiD estimator for catch decreases slightly once these species under rebuilding plans are removed from the sample, but there remains a significantly negative treatment effect due to the SEMRN. The DiD estimator in the GVP regression remains negative and statistically insignificant. This suggests that, while the DiD estimator in the baseline case may be overestimated due to the presence of these rebuilding species, the inclusion of these species does not appear to bias the negative treatment effect observed in Section 5.1.

When the treatment group is comprised of only the SESSF sectors, we find that the treatment effect for catch remains negative and statistically significant, although again, the treatment effect decreases, from 245 to 232 tonnes. However, the treatment effect for GVP is positive in the re-estimated model, while in the baseline model, the effect was negative. The effect remains statistically insignificant. This suggests that the SEMRN may have had a positive effect on the GVP of the SESSF fisheries, and further suggests that the negative result seen in the baseline model was driven by the other treatment fisheries.[31] Finally, when the network is considered to have been established in the 2007/08 financial year, not 2008/09, we find similar results for both catch and GVP, as in the baseline model. The negative treatment effect for catch increases slightly, from 245 to 285 tonnes, while the negative treatment effect for GVP decreases from $323,000 to $198,000.

4.6. Discussion

MPA networks are being established worldwide in response to global commitments to conservation targets, such as the Aichi 11 target of the Convention on Biological Diversity. In addition to achieving conservation benefits, it is anticipated that these MPA networks will generate benefits for marine resource users. The increase in the number of MPA networks around the world means that a better understanding of how these networks can be expected to affect fishery performance is important. However, the literature on *ex post* estimation of the treatment effect of MPA networks on fishery performance is extremely scarce (Ferraro et al., 2018). In this paper, we explore the impacts of MPA networks on adjacent fisheries, using Australia's SEMRN as a case study. We use a panel dataset consisting of eight fisheries, thirty-nine species and fourteen years and apply a difference-in-differences modelling approach to isolate the effect of this network for adjacent fisheries.

Our results show that catch for species in fisheries adjacent to the SEMRN declined in the years subsequent to the establishment of this network. This conclusion is robust to both the removal of species subject to rebuilding plans and fishery management changes that reduced fishing effort and introduced formal harvest strategies, suggesting that the decline in catch is unlikely

[31] This is further suggested when the treatment group is limited to the SBTF. In this case we observe a statistically significant negative treatment effect for both catch and GVP.

to be solely the result of an increase in the amount of precaution exercised in fishery management. There are several reasons why catch may have declined in the region as a result of the MPA network. One reason may be that the conservation benefits anticipated from MPAs, including enhanced biomass and density within the boundaries of both multi-use and no-use MPAs (Sciberras et al., 2015) have not yet eventuated. The SEMRN is less than ten years old, and so the spillover of biomass into adjacent fisheries which may arise as a result of increased density of biomass within MPA boundaries may have not yet occurred (Edgar et al., 2014). This is perhaps reflected in the total allowable catch for species within the treatment group, which has stagnated in the post-2007 era. While the TAC for some species has increased, the overall TAC for species within the treatment group has been relatively stagnant, with quota latency also evident throughout the time period (Patterson et al., 2017; Woodhams et al., 2013). This suggests that increases in biomass, which should be reflected in increases in TAC, have not yet eventuated.

Another reason for the negative treatment effect of the SEMRN on catch may be that behavioural changes observed during the time series, including effort relocation, may have contributed to the negative impact of the SEMRN on adjacent fisheries. Fishers' propensity to adapt to the implementation of no-use MPAs by either searching for new fishing grounds or reallocating effort within available fishing grounds (Horta e Costa et al., 2013), or reducing overall effort in response to closure of fishing grounds (Mason et al., 2012) may lead to declines in catch, particularly in the short-term (Hopf et al., 2016a). For example, fishing effort location in the SESSF gillhook and trap sector drifted over the time series (Appendix 1), and our results shows that the decline in catch in response to the SEMRN persists when the treatment group is limited to these fisheries. The negative treatment effect on catch may, therefore, reflect the fact that the shifts in the location of fishing activity for the gillhook and trap sector of the SESSF, which includes key species such as blue-eye trevalla, gummy shark and sawshark, have contributed to the decline in catch observed in the treatment group as a result of the SEMRN. Although efforts were made to avoid the complete closure of key fishing grounds (Buxton et al., 2006), it may be that the short-term negative consequences generally associated with no-use MPAs, in terms of lost fishing grounds and effort displacement, were not fully ameliorated through implementing multi-use MPAs.

The negative treatment effect evident in our results may also reflect the potential impacts of climate change in the region. The south-east of Australia is subject to an ocean warming ‘hotspot’ (Hobday & Lough, 2011; Hobday et al., 2006; Hobday & Pecl, 2014; Popova et al., 2016), in which ocean temperatures increase at a relatively faster rate in response to climate change than other regions (Pecl et al., 2014). There is evidence of increases in sea surface temperature and an increase in the southward range of the East Australian current (EAC) driven in part by climate change (Ridgway, 2007), and so biological responses to climate change are likely to have occurred prior to the establishment of the SEMRN (Hobday et al., 2006). While the impacts of ocean warming on individual species are highly variable (Harley et al., 2006), impacts for species in the south-east region have included changes in distribution or migration of species (Last et al., 2011; Ling et al., 2009; Ridgway, 2007). Additional impacts may include changes in growth rates, reproductive output and an increased susceptibility to disease (Pecl et al., 2011), all of which may contribute to declines in catch (Brander, 2007). There are two ways in which climate change impacts may influence our estimate of the treatment effect. The first is that, if every species in the treatment group was negatively impacted by ocean warming after the establishment of the SEMRN, this will be incorporated into the treatment effect, and so the negative treatment effect on catch will be overstated. The second is that, given the potential effects MPAs have in hedging against the effects of climate change (Roberts et al., 2017), it is possible that the decline in catch of species in the region would have been greater if not for the MPA network. That is, if every species in the treatment group was negatively impacted by ocean warming, but the SEMRN has acted as a buffer against these adverse impacts, the negative treatment effect will be understated. It is also possible that, if some species in the treatment group were negatively influenced by ocean warming, and others positively influenced, then overall the effects of climate change may not have a significantly confounding effect on our results.

Our results also show no heterogeneity in treatment effects between the treatment group overall and species deemed highly valued. There is no indication the species which were large contributors to GVP prior to the network were more or less affected by the establishment of the SEMRN, beyond the impact which all species in the treatment group were subject to. The avoidance of any specific impact on these key species may help to explain the lack of a significant treatment effect on GVP as a result of the SEMRN, a result which persists for all model specifications considered in this paper. An additional contributing factor to the lack of a treatment effect for GVP may be the increases in the price of a large proportion of species in the treatment group in the post-SEMRN period. While the majority of species in the treatment group declined in value after the network was established, several key species within the SESSF increased in price, including tiger flathead, blue-eye trevalla, gummy shark and silver trevally (van Putten, unpublished data).[32] These species, with the exception of silver trevally, were identified as key commercial species prior to the network being established and which, coupled with the increase in average catch for silver trevally subsequent to the network, may be having a compensatory effect on GVP. In addition to this, the key commercial species in the treatment group identified in the year immediately prior to the network commencing, remained key drivers of GVP in subsequent years, with the exception of blue grenadier. This suggests first, that fishers did not adapt their targeting behaviour in response to the network to the extent

[32] These fish are destined for fish markets in south-east Australia, including the Sydney Fish Market, which supplies to domestic and international buyers.

of changing the key target species in the treatment fisheries, and second, that changes in market conditions in the post-SEMRN period may have compensated for declines in catch, such that we observe a negative treatment effect for catch, but not for GVP.

During the time frame the SEMRN was established, there were two policies introduced that were expected to impact the catch and GVP of fisheries in the sample; the introduction of the harvest strategy policy, which applied to all Commonwealth fisheries, and the vessel buyback scheme, which affected two fisheries in the treatment group and two fisheries in the control group. These two policies, along with the SEMRN, formed part of the Securing Our Fishing Future package introduced in 2005. We find that the treatment effect of the SEMRN is not affected by these policy changes, suggesting that the fisheries management policies introduced as part of the package did not act in synergy with the SEMRN to improve fishery outcomes. This is not an unexpected result, given each instrument had individually specific goals and objectives despite being implemented as part of the same suite of marine management changes. However, it may be that if the MPA network and fisheries management policies had acted in a more cohesive manner that the declines in catch associated with the MPA network could have been avoided.

Several caveats need to be noted when interpreting our results, and more work is needed to further explore empirically the impact of MPA networks on fisheries. First, an issue stemming from our choice of treatment and controls groups is that, while the allocation of the selected fisheries into groups enabled a larger sample size than otherwise, this choice may result in inappropriate comparisons between groups, for example by enabling comparisons between fisheries which experienced changes in fishery management regulations over the time series, which may not be completely captured by the inclusion of fixed effects. One option for future research is to limit the sample to fisheries between which more appropriate comparisons may be made, for example fisheries which share species (or at least share species exhibiting similar behaviours) or use similar fishing gears. Second, our modelling framework does not include any variable other than fixed effects that explicitly control for ecosystem changes associated with climate change. Since the effects of an MPA network is sensitive to climate change impacts on marine ecosystems (Brander, 2007), it is of great interest to disentangle the effects of climate change from the estimation of the average treatment effect of the MPA network. Third, data availability confines our analysis to the use of catch and GVP as proxies for the performance of fisheries. However, there are other possible performance indicators which could be used to indicate the overall performance of a fishery, such as the net economic return of fisheries. Net economic return (NER) is a performance indicator which includes costs as well as value, and so provides an indication of the profitability of a fishery. NER is considered a more appropriate metric to assess fishery performance (White et al., 2008), and so future research which includes performance measures that reflect not only changes in revenue, but also changes in costs, may yield different results to the ones found here. In addition to this, catch and GVP are measures of the commercial value of a fishery. However, the benefits derived from an MPA network go beyond the commercial value of fisheries to include values held by other marine resource users, such as recreational fishers and marine-based tourism (Bhat, 2003; Grafton et al., 2011; Pascoe et al., 2014; Xuan & Armstrong, 2018), as well as non-use values such as existence and bequest values (Mascia, 2004). Incorporating a wider range of indicators when assessing the effects of MPA networks may lead to different conclusions concerning the benefits derived from MPA networks.

Appendix: Fishing effort migration in the SESSF

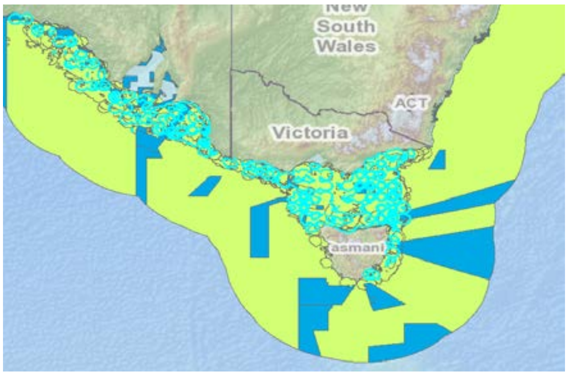
Acknowledgement: Data on relative fishing intensity and the total area fished were supplied by ABARES and were derived from fishery logbook data supplied by the AFMA.

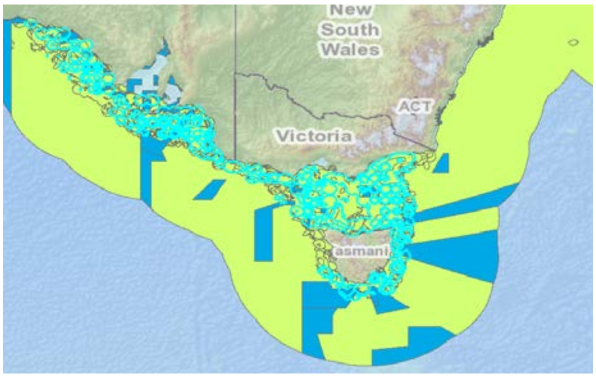
This appendix present maps of fishing effort location for the SESSF gillhook and trap sector and Commonwealth trawl sector. All maps were produced in ArcGIS Desktop 10.5.1. Green area shows the boundary of the SESSF fishery.[33] Dark blue area shows the boundaries of the SEMRN.[34] Light blue lines show the location of fishing activity, for each subsector/sector in a given year.

[33] According to the Fishery Management Regulations Act 1992 (Cth). Data retrieved from Geoscience Australia (https://data.gov.au/dataset/commonwealth-fisheries-2006) 30 August 2018.
[34] Note the SEMRN was not established until 2007 but is included in all figures for consistency.

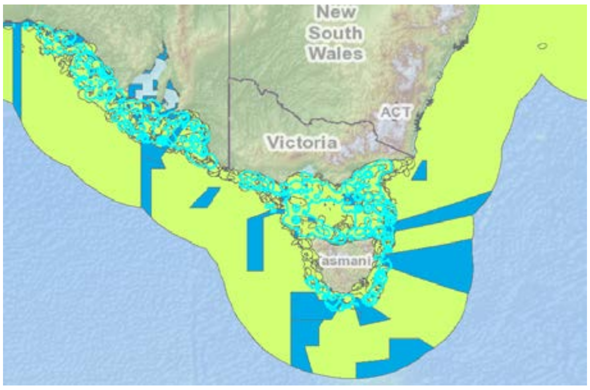
Figure A1. Fishing activity (measured in length of net deployed) for the shark gillnet subsector of the gillhook and trap sector between 2001 and 2015.

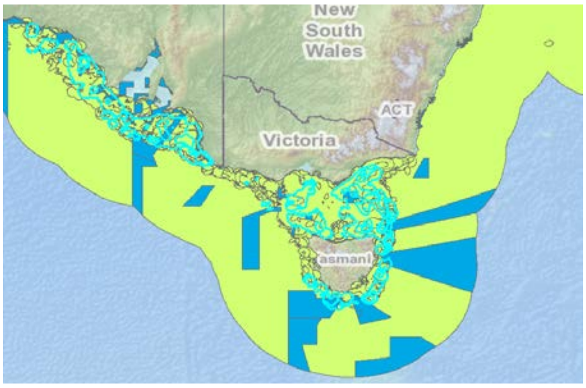
2001 **2003**



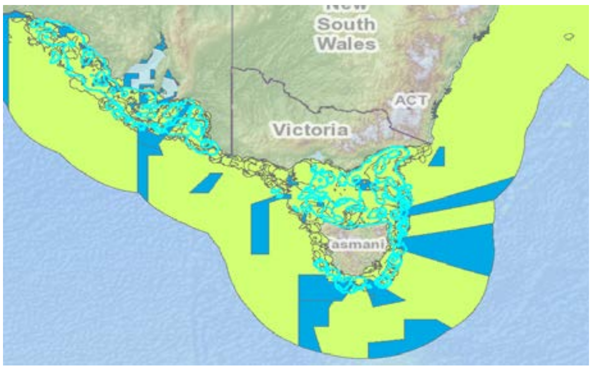


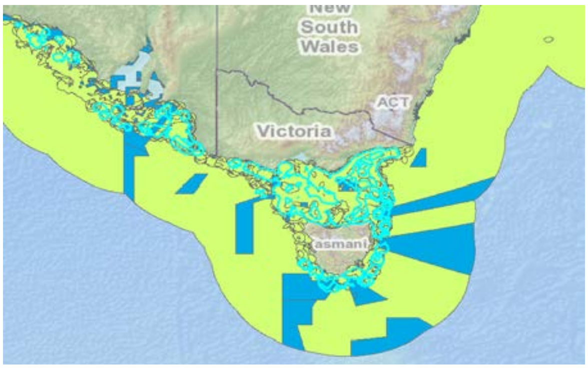
2005 **2007**



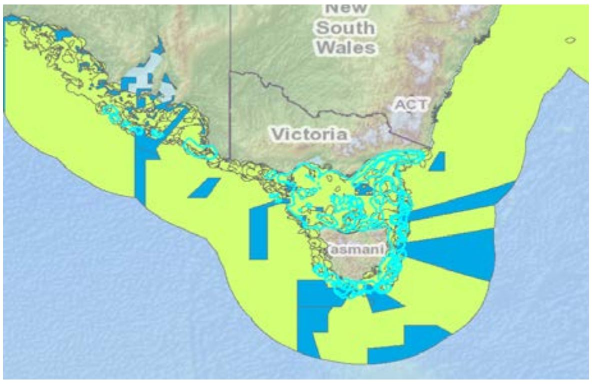


2009 **2011**





2013 **2015**



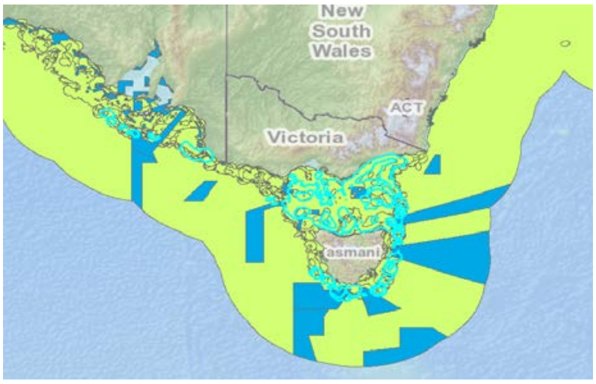
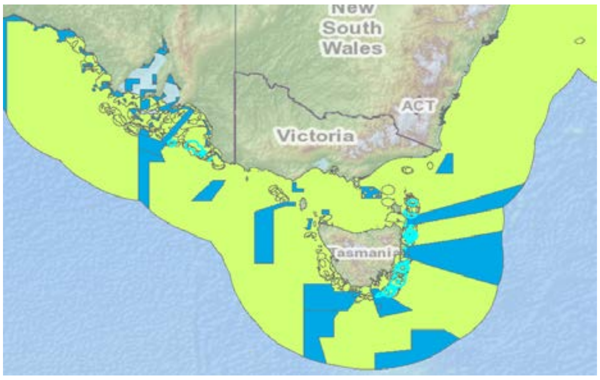
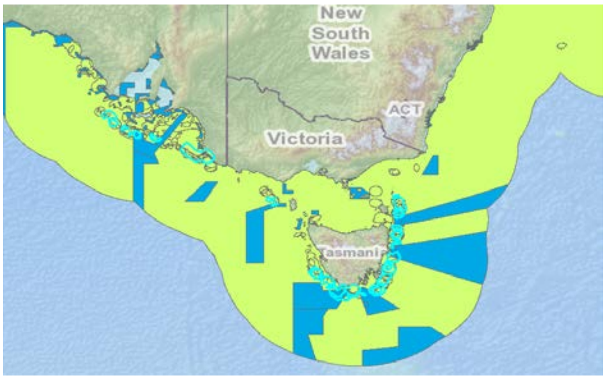


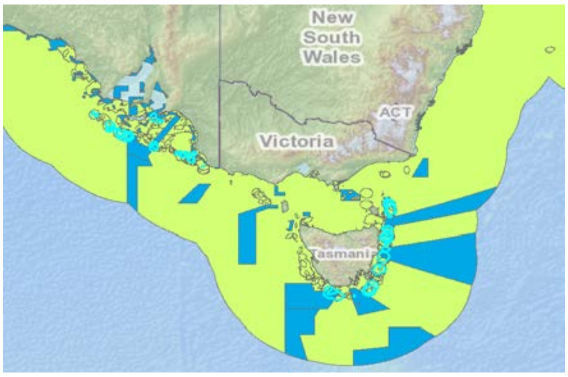
Figure A2. Fishing activity (measured in number of hooks deployed per operation) for the sharkhook and scalehook subsector of the gillhook and trap sector between 2001 and 2015.

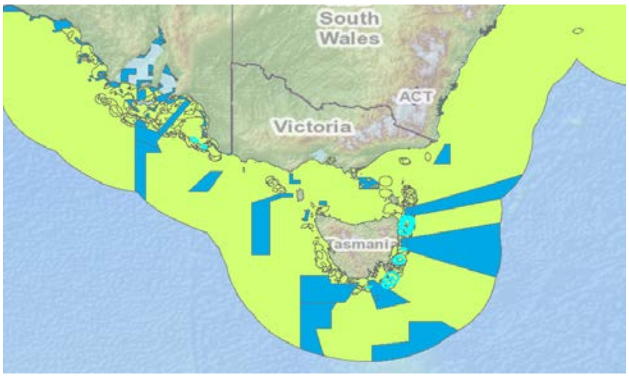
2001 **2003**



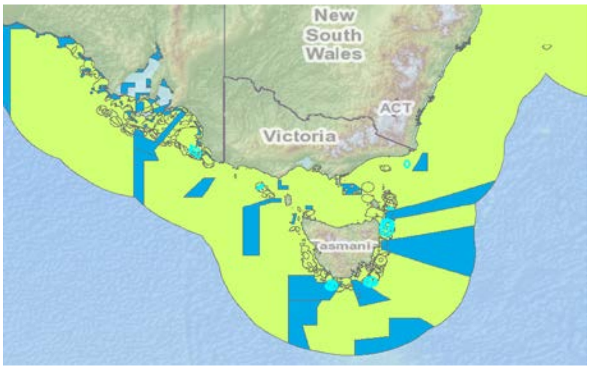


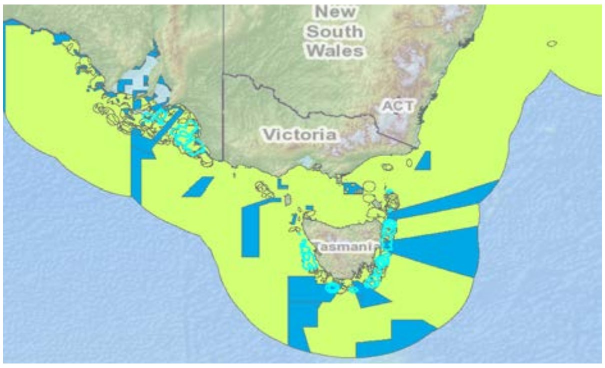
2005 **2007**



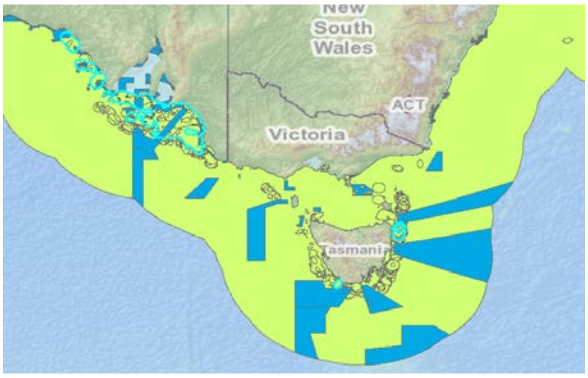


2009 **2011**





2013 **2015**



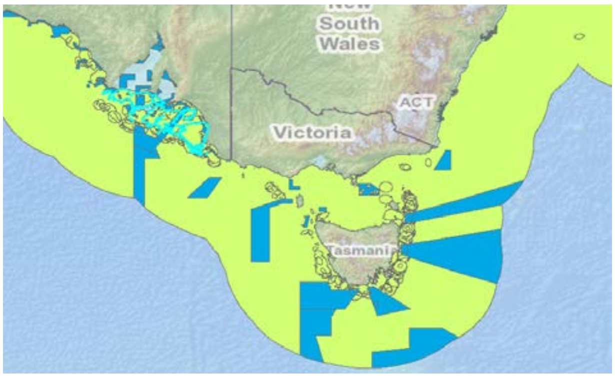
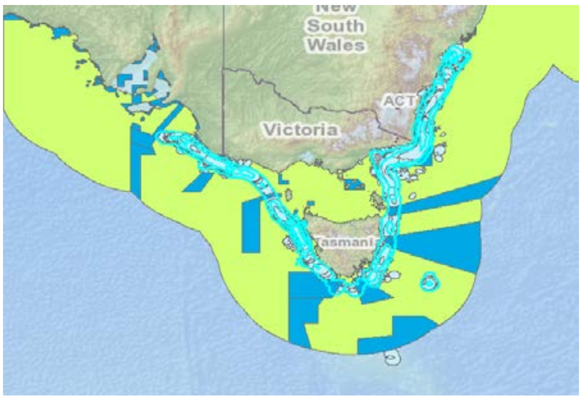
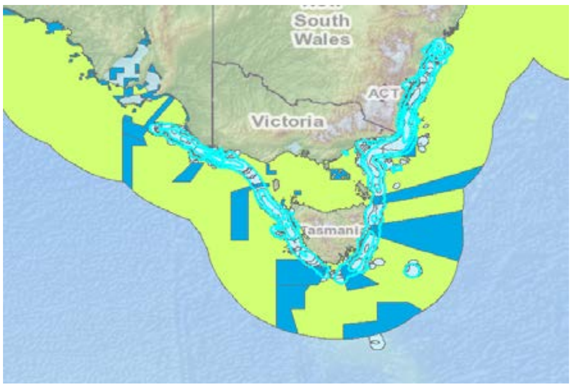


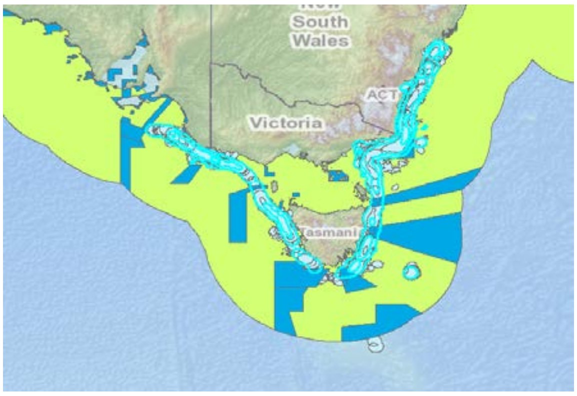
Figure A3. Fishing activity (measured in number of hours trawled) for the Commonwealth trawl sector between 2001 and 2015.

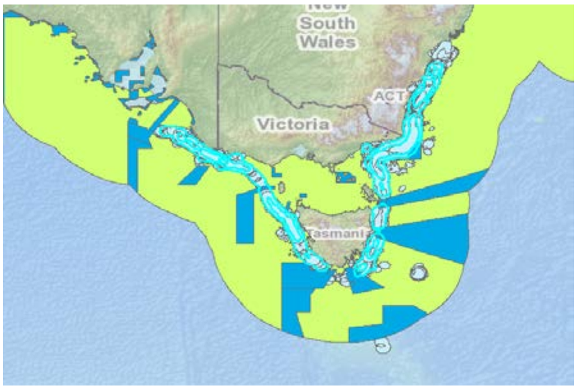
2001 **2003**





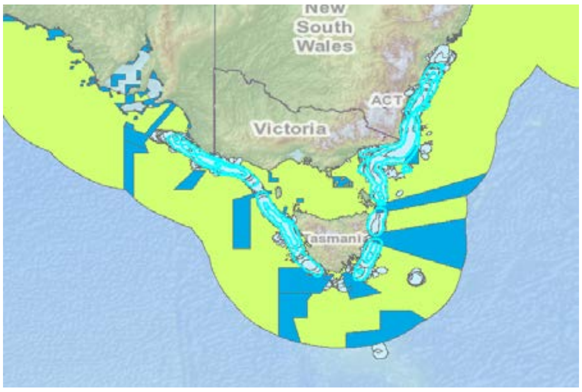
2005 **2007**

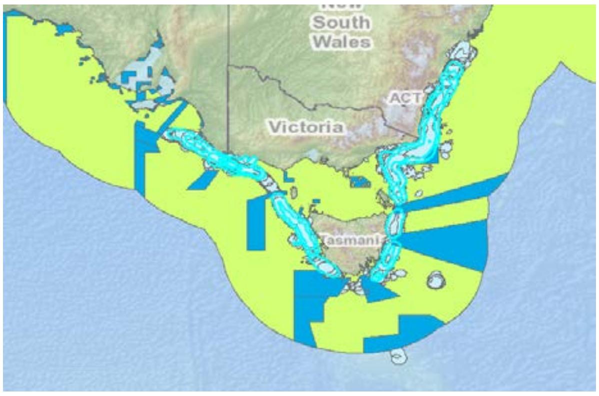




2009

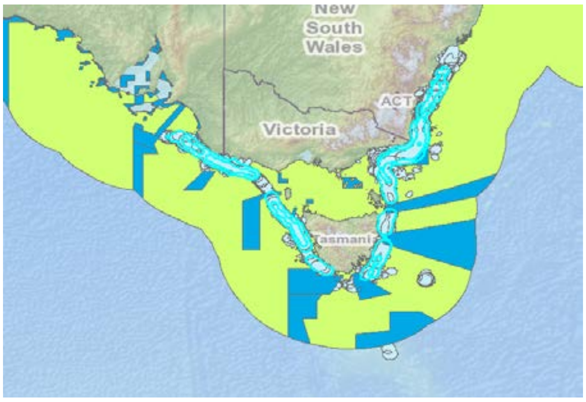
2011

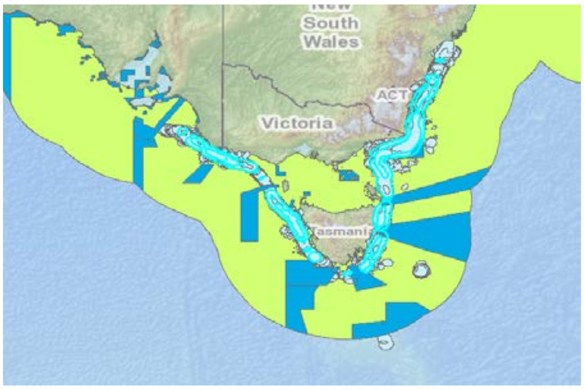




2013

2015





Chapter 5: Thesis Conclusion

The problem of sustainably managing fishery resources is complex, given the myriad relationships and connections between the biological realities of the fish stocks and the ecosystems they inhabit, and the conflicting needs and desires of resource users. This problem is made even more complex when fishing activity results in habitat damage. This thesis has contributed to our understanding of this problem by examining: first, how fishing-induced habitat degradation impacts the application of the precautionary principle when attempting to rebuild fish stocks, second, the role of these habitat-fishery linkages when making decisions concerning catch share allocations, and third, how mechanisms designed to improve habitat can influence the catch and revenue of fisheries.

This thesis has demonstrated that for a range of circumstances, the relationships between habitat and fishing activity are such that one cannot be managed without having consequences for the other, and so information relating to both should be included jointly in decision making. This thesis finds that omitting information about the environmental impacts of fishing activity when making decisions concerning the exploitation of fish stocks can have destructive consequences for the sustainability of fish populations. The first essay develops a bioeconomic model where fishing-induced habitat degradation presents through the carrying capacity of the fish population, which is negatively impacted by fishing activity but is capable of regenerating over time. This essay shows that, relative to where no habitat-fishery linkages exist, a failure to include information about these linkages when formulating stock recovery plans may increase the risk that a fish stock will take longer to rebuild to a target level of biomass or will fail to rebuild altogether. This risk decreases when no-use MPAs are used as a means of controlling fishing mortality alongside a harvest control rule, as the MPA not only protects the biomass from harvest but also counteracts the negative feedback imposed on the fish stock biomass through habitat degradation. Additionally, using no-use MPAs in concert with harvest control rules in stock recovery plans can have the benefit of reducing or removing the trade-offs between different fishery performance indicators where habitat impacts exist. This is because applying the precautionary principle through a no-use MPA means that less precaution need be applied through the harvest control rule. The first essay also highlights that the duration of the stock rebuilding period decreases when a no-use MPA is implemented as part of a stock recovery plan, regardless of whether habitat impacts result from fishing activity. This suggests that the use of the two control mechanisms in conjunction promotes a more efficient stock recovery irrespective of the habitat effect, although this efficiency is heightened where habitat effects are apparent.

The second essay further exposes the negative impacts of excluding information about fishing-induced habitat degradation when making decisions concerning the economically optimal level of total allowable catch and its allocation across different resource user groups in a fishery. This essay also uses a bioeconomic model in which fishing-induced habitat degradation presents through the carrying capacity of the fish population. This essay finds that a failure to account for the impacts of fishing on the habitat may lead to inadvertent stock collapse, overfishing and declines in the profitability of the fishery, particularly in circumstances where the fishing gear used is destructive, or where the habitat is slow to recover from the inflicted damage. This essay also shows that including information about the habitat effect may lead to the economically optimal exclusion of users from the fishery, either because of excessive habitat damage in vulnerable environments or the relatively higher cost of fishing in resilient environments. The decision to allocate catch shares inconsistent with the optimal allocation, either for social or political reasons, will lead to trade-offs between fishery performance indicators. The extent of these trade-offs is, however, highly dependent on the initial allocations made, the resilience of the environment and stock to habitat damage, and the rate of fishing-induced habitat damage. Overall, this essay shows that improved understanding of when information concerning damaging interactions between fish stock habitats and fishing activity ought to be included in the harvest allocation process will aid fishery managers in making decisions concerning data gathering and ecological risk assessment for fishing activity.

The third essay of this thesis investigates the role of habitat conservation in generating benefits for fisheries, by examining whether a network of MPAs designed to conserve the marine habitat led to any discernible improvements in performance for adjacent fisheries in the Australian context. One of the justifications for establishing networks of MPAs, rather than isolated MPAs, is that a network permits more flexibility in its implementation to ensure fisheries are not negatively impacted, for example through lost access to fishing grounds, and have more scope for delivering benefits to adjacent fisheries in the form of increased catch rates. Here we find evidence that the network of MPAs established in south-east Australia in 2007 had a negative impact on catch for the adjacent fisheries, but that this decline in catch did not correspond to a fall in the gross value of production. This result is robust to controlling for fisheries management changes which occurred at the same time. Additionally, no heterogeneity in impact is found between commercially valued species and for all impacted species. This suggests that, while a network of MPAs may not generate benefits to adjacent fisheries, it may be possible to at least avoid negative impacts for specific species, such as by taking advantage of the flexibility of MPA network zoning to minimise adverse impacts on species which are key drivers of fishery performance.

Overall, this thesis has shown that the negative connections between fishing activity and habitat may undermine the sustainability of fishery resources by compromising rebuilding objectives and contributing to the overfishing of fish stocks. Further, this thesis finds that tools traditionally used for habitat conservation may not generate benefits to adjacent fisheries where established solely as a tool for habitat management, even where consideration is given to adjacent fisheries before implementation. Therefore, while it seems that fishery management in the future may benefit from the inclusion of a wider range of information concerning ecosystem characteristics, including the vulnerability of habitats to fishing activity. Additionally, while fishery management may benefit from habitat management tools such as no-use MPAs being used in conjunction with fishery management tools to promote the sustainable exploitation of fishery resources, it seems that MPAs alone may be unable to generate benefits for fisheries. However, the research undertaken in this thesis is subject to several caveats and limitations, each of which provide avenues for future research.

First, the bioeconomic analyses in the first two essays focus on fishing-induced habitat degradation that presents via the carrying capacity of the fish population. However, the adverse impacts of habitat damage on the productivity of stocks may occur through different channels, for example through the intrinsic growth rate of the population. Exploring the consequences of alternate manifestations of habitat degradation for stock recovery plans and allocations of catch shares between user groups may be a useful extension of the work done in this thesis. Second, this thesis has characterised fishery performance predominately in terms of catch, total revenue and overall profits. This characterisation ignores a range of different aspects of fishery performance, as well as disregarding the needs of other resource users. For example, the benefits derived from no-use MPAs and MPA network go beyond the commercial value of fisheries to include values produced by activities such as marine-based tourism, as well as generating non-use values such as existence and bequest values. Additionally, the assumption that benefit to fishery users may be measured in terms of economic value may be valid for commercial users, but not for other users such as recreational and Indigenous fishers who derive cultural, health and livelihood benefits from fishing. Considering alternative measures of fishery performance and marine user welfare may lead to more comprehensive conclusions concerning the management of fishery resources in the presence of fishing-induced habitat damage. Third, this thesis has not incorporated information about the spatial characteristics of fisheries or marine habitats, such as geographic location of fishing effort and the behavioural responses of fishers over time, the distribution of fish populations and marine habitats or the dispersion of climate change impacts. Including information about these characteristics may lead to different conclusions concerning the benefits generated by MPA networks for adjacent fisheries, and the efficacy of incorporating no-use MPAs into stock recovery plans. Finally, the empirical examination in the third essay involved the placement of species into treatment and control groups. Allocation of species was made to ensure the largest sample size possible, although this choice may result in inappropriate comparisons between groups, for example between fisheries which do not share common management features or habitat characteristics. One option for future research is to focus on comparisons between fisheries which share common species (or at least share species exhibiting similar behaviours), use similar fishing gears, share key management features such as individual transferable quotas (ITQs) or seasonal closures, or have common habitat features.

Bibliography

Abesamis, R. A., & Russ, G. R. (2005). DENSITY-DEPENDENT SPILLOVER FROM A MARINE RESERVE: LONG-TERM EVIDENCE. *Ecological Applications, 15*(5), 1798-1812. doi:10.1890/05-0174

Acott, T. G., & Urquhart, J. (2014). Sense of Place and Socio-cultural Values in Fishing Communities Along the English Channel. In J. Urquhart, T. G. Acott, D. Symes, & M. Zhao (Eds.), *Social Issues in Sustainable Fisheries Management* (pp. 257-277). Dordrecht: Springer Netherlands.

Agardy, T., Bridgewater, P., Crosby, M. P., Day, J., Dayton, P. K., Kenchington, R., . . . Peau, L. (2003). Dangerous targets? Unresolved issues and ideological clashes around marine protected areas. *Aquatic Conservation: Marine and Freshwater Ecosystems, 13*(4), 353-367. doi:10.1002/aqc.583

Agardy, T., di Sciara, G. N., & Christie, P. (2011). Mind the gap: Addressing the shortcomings of marine protected areas through large scale marine spatial planning. *Marine Policy, 35*(2), 226-232. doi:http://dx.doi.org/10.1016/j.marpol.2010.10.006

Airoldi, L., Balata, D., & Beck, M. W. (2008). The Gray Zone: Relationships between habitat loss and marine diversity and their applications in conservation. *Journal of Experimental Marine Biology and Ecology, 366*(1), 8-15. doi:https://doi.org/10.1016/j.jembe.2008.07.034

Akpalu, W., & Bitew, W. (2014). Optimum reserve size, fishing induced change in carrying capacity, and phenotypic diversity. *Journal of Bioeconomics, 16*(3), 289-304. doi:10.1007/s10818-014-9178-8

Akpalu, W., & Bitew, W. T. (2011). Species diversity, fishing induced change in carrying capacity and sustainable fisheries management. *Ecological Economics, 70*(7), 1336-1343. doi:10.1016/j.ecolecon.2011.02.008

Almany, G. R., Connolly, S. R., Heath, D. D., Hogan, J. D., Jones, G. P., McCook, L. J., . . . Williamson, D. H. (2009). Connectivity, biodiversity conservation and the design of marine reserve networks for coral reefs. *Coral Reefs, 28*(2), 339-351. doi:10.1007/s00338-009-0484-x

Anderies, J. M., Janssen, M. A., & Ostrom, E. (2004). A Framework to Analyze the Robustness of Social-ecological Systems from an Institutional Perspective. *Ecology and Society, 9*(1), 18.

ANZECC-TFMPA. (1998). Guidelines for Establishing the National Representative System of Marine Protected Areas. *Australian and New Zealand Environment and Conservation Council, Task Force on Marine Protected Areas, Environment Australia,Canberra*.

Arbo, P., & Thủy, P. T. T. (2016). Use conflicts in marine ecosystem-based management — The case of oil versus fisheries. *Ocean & Coastal Management,* *122*(Supplement C), 77-86. doi:https://doi.org/10.1016/j.ocecoaman.2016.01.008

Arias, A., Cinner, J. E., Jones, R. E., & Pressey, R. L. (2015). Levels and drivers of fishers' compliance with marine protected areas. . *Ecology and Society, 20*(4), 19.

Armstrong, C. W. (2007). A note on the ecological–economic modelling of marine reserves in fisheries. *Ecological Economics, 62*(2), 242-250. doi:10.1016/j.ecolecon.2006.03.027

Armstrong, C. W., & Falk-Petersen, J. (2008). Habitat–fisheries interactions: a missing link? *ICES Journal of Marine Science: Journal du Conseil, 65*(6), 817-821. doi:10.1093/icesjms/fsn092

Armstrong, C. W., Kahui, V., Vondolia, G. K., Aanesen, M., & Czajkowski, M. (2017). Use and Non-Use Values in an Applied Bioeconomic Model of Fisheries and Habitat Connections. *Marine Resource Economics, 32*(4), 351-369. doi:10.1086/693477

Armstrong, C. W., & Sumaila, U. R. (2000). Cannibalism and the Optimal Sharing of the North-East Atlantic Cod Stock: a Bioeconomic Model. *Journal of Bioeconomics, 2*(2), 99-115. doi:10.1023/a:1011429907162

Arnason, R. (1990). Minimum Information Management in Fisheries. *The Canadian Journal of Economics / Revue canadienne d'Economique, 23*(3), 630-653. doi:10.2307/135652

Arnason, R. (2008). Conflicting uses of marine resources: can ITQs promote an efficient solution?*. *Australian Journal of Agricultural and Resource Economics, 53*(1), 145-174. doi:10.1111/j.1467-8489.2007.00424.x

Arnason, R. (2012). Property Rights in Fisheries: How Much Can Individual Transferable Quotas Accomplish? *Review of Environmental Economics and Policy, 6*(2), 217-236. doi:10.1093/reep/res011

Ayer, A., Fulton, S., Caamal-Madrigal, J. A., & Espinoza-Tenorio, A. (2018). Halfway to sustainability: Management lessons from community-based, marine no-take zones in the Mexican Caribbean. *Marine Policy, 93*, 22-30. doi:https://doi.org/10.1016/j.marpol.2018.03.008

Ballantine, B. (2014). Fifty years on: Lessons from marine reserves in New Zealand and principles for a worldwide network. *Biological Conservation, 176*, 297-307. doi:https://doi.org/10.1016/j.biocon.2014.01.014

Bennett, N. J., Kaplan-Hallam, M., Augustine, G., Ban, N., Belhabib, D., Brueckner-Irwin, I., . . . Bailey, M. (2018). Coastal and Indigenous community access to marine resources and the ocean: A policy imperative for Canada. *Marine Policy, 87*, 186-193. doi:https://doi.org/10.1016/j.marpol.2017.10.023

Berkes, F. (2011). Restoring Unity. In R. E. O. T. J. Pitcher, R. I. Perry, K. Cochrane and P. Cury (Ed.), *World Fisheries*.

Bhat, M. G. (2003). Application of non-market valuation to the Florida Keys marine reserve management. *J Environ Manage, 67*(4), 315-325. doi:http://dx.doi.org/10.1016/S0301-4797(02)00207-4

Boncoeur, J., ALBAN, F., GUYADER, O., & THÉBAUD, O. (2002). FISH, FISHERS, SEALS AND TOURISTS: ECONOMIC CONSEQUENCES OF CREATING A MARINE RESERVE IN A MULTI - SPECIES, MULTI - ACTIVITY CONTEXT. *Natural Resource Modeling, 15*(4), 387-411. doi:doi:10.1111/j.1939-7445.2002.tb00095.x

Brander, K. M. (2007). Global fish production and climate change. *Proceedings of the National Academy of Sciences, 104*(50), 19709-19714. doi:10.1073/pnas.0702059104

Brown, C. J., Abdullah, S., & Mumby, P. J. (2015). Minimizing the Short-Term Impacts of Marine Reserves on Fisheries While Meeting Long-Term Goals for Recovery. *Conservation Letters, 8*(3), 180-189. doi:doi:10.1111/conl.12124

Buxton, C., Haddon, M., & Bradshaw, M. (2006). *Regional Impact Assessment for the Marine Protected Areas proposed for the South-East Region*.

Buxton, C. D., Hartmann, K., Kearney, R., & Gardner, C. (2014). When Is Spillover from Marine Reserves Likely to Benefit Fisheries? *PLoS ONE, 9*(9), e107032. doi:10.1371/journal.pone.0107032

Cabral, R. B., Gaines, S. D., Johnson, B. A., Bell, T. W., & White, C. (2017). Drivers of redistribution of fishing and non-fishing effort after the implementation of a marine protected area network. *Ecological Applications, 27*(2), 416-428. doi:doi:10.1002/eap.1446

Caddy, J. F., & Agnew, D. J. (2004). An overview of recent global experience with recovery plans for depleted marine resources and suggested guidelines for recovery planning. *Reviews in Fish Biology and Fisheries, 14*(1), 43-112. doi:10.1007/s11160-004-3770-2

Cadrin, S. X., & Dickey-Collas, M. (2015). Stock assessment methods for sustainable fisheries. *ICES Journal of Marine Science, 72*(1), 1-6. doi:10.1093/icesjms/fsu228

Cadrin, S. X., & Pastoors, M. A. (2008). Precautionary harvest policies and the uncertainty paradox. *Fisheries Research, 94*(3), 367-372. doi:http://dx.doi.org/10.1016/j.fishres.2008.06.004

CAPAD. (2008). Collaborative Australian Protected Area Database - CAPAD 2008 - External. . *Department of Environment, Water, Heritage and Arts, Canberra.*

Card, D., & Krueger, A. (1992). Does School Quality Matter? Returns to Education and the Characteristics of Public Schools in the United States. *Journal of Political Economy, 100*(1), 1-40.

Caveen, A., Polunin, N., Gray, T., & Stead, S. (2015). *The Controversy over Marine Protected Areas: Science meets Policy*. New York: Springer.

CBD. (2011). Aichi Target 11. Decision X/2. In S. o. t. CBD (Ed.): Convention on Biological Diversity.

Charles, A., & Wilson, L. (2009). Human dimensions of Marine Protected Areas. *ICES Journal of Marine Science, 66*(1), 6-15. doi:10.1093/icesjms/fsn182

Chollett, I., Box, S. J., & Mumby, P. J. (2016). Quantifying the squeezing or stretching of fisheries as they adapt to displacement by marine reserves. *Conservation Biology, 30*(1), 166-175. doi:doi:10.1111/cobi.12573

Clark, C. W. (1990). *Mathematical Bioeconomics: Optimal Management of Renewable Resources* (2nd ed.): John Wiley & Sons.
Clark, C. W., Clarke, F. H., & Munro, G. R. (1979). The Optimal Exploitation of Renewable Resource Stocks: Problems of Irreversible Investment. *Econometrica, 47*(1), 25-47. doi:10.2307/1912344
Clark, C. W., & Munro, G. R. (1975). The economics of fishing and modern capital theory: A simplified approach. *Journal of Environmental Economics and Management, 2*(2), 92-106.
Conrad, J. (1999). The Bioeconomics of Marine Sanctuaries. *Journal of Bioeconomics, 1*(2), 205-217. doi:10.1023/A:1010039031324
Costello, C., Gaines, S. D., & Lynham, J. (2008). Can Catch Shares Prevent Fisheries Collapse? *Science, 321*(5896), 1678-1681. doi:10.1126/science.1159478
Crowe, F. M., Longson, I. G., & Joll, L. M. (2013). Development and implementation of allocation arrangements for recreational and commercial fishing sectors in Western Australia. *Fisheries Management and Ecology, 20*(2-3), 201-210. doi:10.1111/j.1365-2400.2012.00858.x
Day, J., Dudley, N., Hockings, M., Holmes, G., Laffoley, D., Stolton, S., & Wells, S. (2012). Guidelines for applying the IUCN Protected Area Management Categories to Marine Protected Areas. . *Gland, Switzerland: IUCN. 36pp.*
Dayton, P. K., Thrush, S. F., Agardy, M. T., & Hofman, R. J. (1995). Environmental effects of marine fishing. *Aquatic Conservation: Marine and Freshwater Ecosystems, 5*(3), 205-232. doi:10.1002/aqc.3270050305
De Leo, G. A., & Micheli, F. (2015). The good, the bad and the ugly of marine reserves for fishery yields. *Philosophical Transactions of the Royal Society B: Biological Sciences, 370*(1681), 20140276. doi:10.1098/rstb.2014.0276
Dichmont, C. M., Pascoe, S., Kompas, T., Punt, A. E., & Deng, R. (2010). On implementing maximum economic yield in commercial fisheries. *Proc Natl Acad Sci U S A, 107*(1), 16-21. doi:10.1073/pnas.0912091107
Dobson, A., Lodge, D., Alder, J., Cumming, G. S., Keymer, J., McGlade, J., . . . Xenopoulos, M. A. (2006). Habitat loss, trophic collapse, and the decline of ecosystem services. *Ecology, 87*(8), 1915-1924.
Dowling, N. A., Dichmont, C. M., Venables, W., Smith, A. D. M., Smith, D. C., Power, D., & Galeano, D. (2013). From low- to high-value fisheries: Is it possible to quantify the trade-off between management cost, risk and catch? *Marine Policy, 40*, 41-52. doi:https://doi.org/10.1016/j.marpol.2012.12.009
Dunne, R. P., Polunin, N. V. C., Sand, P. H., & Johnson, M. L. (2014). Chapter Three - The Creation of the Chagos Marine Protected Area: A Fisheries Perspective☆. In M. L. Johnson & J. Sandell (Eds.), *Advances in Marine Biology* (Vol. 69, pp. 79-127): Academic Press.
Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., . . . Thomson, R. J. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature, 506*, 216.

doi:10.1038/nature13022 https://www.nature.com/articles/nature13022#supplementary-information
Elliott, M., & Cutts, N. D. (2004). Marine habitats: loss and gain, mitigation and compensation. *Marine Pollution Bulletin, 49*(9–10), 671-674. doi:https://doi.org/10.1016/j.marpolbul.2004.08.018
FAO. (1995). Code of Conduct for Responsible Fisheries. *Rome, FAO. 41 p.*
FAO. (1996). Precautionary approach to capture fisheries and species introductions. *FAO Technical Guidelines for Responsible Fisheries. No. 2. Rome, FAO.*
FAO. (2003). The ecosystem approach to fisheries. *FAO Technical Guidelines for Responsible Fisheries. No. 2. Rome, FAO. No. 4, Suppl. 2. Rome, FAO. 112 p.*
FAO. (2016). The State of World Fisheries and Aquaculture 2016. Contributing to food security and nutrition for all. Rome. 200 pp.
FAO. (2018). The State of World Fisheries and Aquaculture 2018 - Meeting the sustainable development goals. Rome. .
Ferraro, P. J., Sanchirico, J. N., & Smith, M. D. (2018). Causal inference in coupled human and natural systems. *Proceedings of the National Academy of Sciences*. doi:10.1073/pnas.1805563115
Fina, M. (2011). Evolution of Catch Share Management: Lessons from Catch Share Management in the North Pacific. *Fisheries, 36*(4), 164-177. doi:10.1080/03632415.2011.564509
Flood, M. J., Stobutzki, I., Andrews, J., Ashby, C., Begg, G. A., Fletcher, R., . . . Wise, B. S. (2016). Multijurisdictional fisheries performance reporting: How Australia's nationally standardised approach to assessing stock status compares. *Fisheries Research, 183*, 559-573. doi:https://doi.org/10.1016/j.fishres.2016.02.002
Foley, N. S., Armstrong, C. W., Kahui, V., Mikkelsen, E., & Reithe, S. (2012). A Review of Bioeconomic Modelling of Habitat-Fisheries Interactions. *International Journal of Ecology, 2012*, 1-11. doi:10.1155/2012/861635
Folke, C. (2007). Social–ecological systems and adaptive governance of the commons. *Ecological Research, 22*(1), 14-15. doi:10.1007/s11284-006-0074-0
Fulton, E. A., Smith, A. D. M., Smith, D. C., & van Putten, I. E. (2011). Human behaviour: the key source of uncertainty in fisheries management. *Fish and Fisheries, 12*(1), 2-17. doi:10.1111/j.1467-2979.2010.00371.x
Gaines, S. D., Gaylord, B., & Largier, J. L. (2003). AVOIDING CURRENT OVERSIGHTS IN MARINE RESERVE DESIGN. *Ecological Applications, 13*(sp1), 32-46. doi:doi:10.1890/1051-0761(2003)013[0032:ACOIMR]2.0.CO;2
Gaines, S. D., White, C., Carr, M. H., & Palumbi, S. R. (2010). Designing marine reserve networks for both conservation and fisheries management. *Proc Natl Acad Sci U S A, 107*(43), 18286-18293. doi:10.1073/pnas.0906473107
Garcia, S. M. (1994). The Precautionary Principle: its implications in capture fisheries management. *Ocean & Coastal Management, 22*(2), 99-125. doi:https://doi.org/10.1016/0964-5691(94)90014-0
Garcia, S. M., & Rosenberg, A. A. (2010). Food security and marine capture fisheries: characteristics, trends, drivers and future perspectives. *Philosophical*

Transactions of the Royal Society B: Biological Sciences, 365(1554), 2869-2880. doi:10.1098/rstb.2010.0171
Gell, F. R., & Roberts, C. M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *Trends in Ecology & Evolution, 18*(9), 448-455. doi:https://doi.org/10.1016/S0169-5347(03)00189-7
Ginter, J. J. C. (1995). The Alaska community development quota fisheries management program. *Ocean & Coastal Management, 28*(1), 147-163. doi:http://dx.doi.org/10.1016/0964-5691(95)00069-0
Girardin, R., Hamon, K. G., Pinnegar, J., Poos, J. J., Thébaud, O., Tidd, A., . . . Marchal, P. (2016). Thirty years of fleet dynamics modelling using discrete-choice models: What have we learned? *Fish and Fisheries*, n/a-n/a. doi:10.1111/faf.12194
Goñi, R., Hilborn, R., Díaz, D., Mallol, S., & Adlerstein, S. (2010). Net contribution of spillover from a marine reserve to fishery catches. *Marine Ecology Progress Series, 400*, 233-243.
González-Laxe, F. (2005). The precautionary principle in fisheries management. *Marine Policy, 29*(6), 495-505. doi:http://dx.doi.org/10.1016/j.marpol.2004.09.002
Gordon, H. S. (1954). The Economic Theory of a Common-Property Resource: The Fishery. *Journal of Political Economy, 62*(2), 124-142.
Grafton, R. Q., Akter, S., & Kompas, T. (2011). A Policy-Enabling Framework for the Ex-Ante Evaluation of Marine Protected Areas. *Ocean & Coastal Management, 54*(6), 478-487. doi:10.1016/j.ocecoaman.2011.03.006
Grafton, R. Q., Kompas, T., & Ha, P. V. (2009). Cod Today and None Tomorrow: The Economic Value of a Marine Reserve. *Land Economics, 85*(3), 454-469.
Grafton, R. Q., Kompas, T., & Lindenmayer, D. (2005). Marine reserves with ecological uncertainty. *Bull Math Biol, 67*(5), 957-971. doi:10.1016/j.bulm.2004.11.006
Grafton, R. Q., Kompas, T., & Pham Van, H. (2006). The Economic Payoffs from Marine Reserves: Resource Rents in a Stochastic Environment*. *Economic Record, 82*(259), 469-480.
Grieve, C. (2009). Environmental and Social Criteria for Allocating Access to Fisheries Resources. *Meridian Prime*.
Gutiérrez, N. L., Hilborn, R., & Defeo, O. (2011). Leadership, social capital and incentives promote successful fisheries. *Nature, 470*, 386. doi:10.1038/nature09689 https://www.nature.com/articles/nature09689#supplementary-information
Hall-Spencer, J., Allain, V., & Fosså, J. H. (2002). Trawling damage to Northeast Atlantic ancient coral reefs. *Proceedings of the Royal Society B: Biological Sciences, 269*(1490), 507-511. doi:10.1098/rspb.2001.1910
Halpern, B. S., Lester, S. E., & McLeod, K. L. (2010). Placing marine protected areas onto the ecosystem-based management seascape. *Proc Natl Acad Sci U S A, 107*(43), 18312-18317. doi:10.1073/pnas.0908503107

Hannesson, R. (1998). Marine Reserves: What Would They Accomplish? *Marine Resource Economics, 13*(3), 159-170. doi:10.2307/42629231

Hardin, G. (1968). The Tragedy of the Commons. *Science, 162*(3859), 1243-1248. doi:10.1126/science.162.3859.1243

Harley, C. D. G., Randall Hughes, A., Hultgren, K. M., Miner, B. G., Sorte, C. J. B., Thornber, C. S., . . . Williams, S. L. (2006). The impacts of climate change in coastal marine systems. *Ecology Letters, 9*(2), 228-241. doi:doi:10.1111/j.1461-0248.2005.00871.x

Harrison, Hugo B., Williamson, David H., Evans, Richard D., Almany, Glenn R., Thorrold, Simon R., Russ, Garry R., . . . Jones, Geoffrey P. (2012). Larval Export from Marine Reserves and the Recruitment Benefit for Fish and Fisheries. *Current Biology, 22*(11), 1023-1028. doi:https://doi.org/10.1016/j.cub.2012.04.008

Haynie, A. C. (2014). Changing usage and value in the Western Alaska Community Development Quota (CDQ) program. *Fisheries Science, 80*(2), 181-191. doi:10.1007/s12562-014-0723-0

Hiddink, J. G., Jennings, S., Sciberras, M., Szostek, C. L., Hughes, K. M., Ellis, N., . . . Kaiser, M. J. (2017). Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. *Proceedings of the National Academy of Sciences*. doi:10.1073/pnas.1618858114

Hiddink, J. G., Johnson, A. F., Kingham, R., & Hinz, H. (2011). Could our fisheries be more productive? Indirect negative effects of bottom trawl fisheries on fish condition. *Journal of Applied Ecology, 48*(6), 1441-1449. doi:10.1111/j.1365-2664.2011.02036.x

Hilborn, R. (2007a). Defining success in fisheries and conflicts in objectives. *Marine Policy, 31*(2), 153-158. doi:https://doi.org/10.1016/j.marpol.2006.05.014

Hilborn, R. (2007b). Managing fisheries is managing people: what has been learned? *Fish and Fisheries, 8*(4), 285-296. doi:10.1111/j.1467-2979.2007.00263_2.x

Hilborn, R., Maguire, J.-J., Parma, A. M., & Rosenberg, A. A. (2001). The Precautionary Approach and risk management: can they increase the probability of successes in fishery management? *Canadian Journal of Fisheries and Aquatic Sciences, 58*(1), 99-107. doi:10.1139/f00-225

Hilborn, R., Micheli, F., & De Leo, G. A. (2006). Integrating marine protected areas with catch regulation. *Canadian Journal of Fisheries and Aquatic Sciences, 63*(3), 642-649. doi:10.1139/f05-243

Hilborn, R., Stokes, K., Maguire, J.-J., Smith, T., Botsford, L. W., Mangel, M., . . . Walters, C. (2004). When can marine reserves improve fisheries management? *Ocean & Coastal Management, 47*(3-4), 197-205. doi:10.1016/j.ocecoaman.2004.04.001

Hobday, A. J., & Lough, J. M. (2011). Projected climate change in Australian marine and freshwater environments. *Marine and Freshwater Research, 62*(9), 1000-1014. doi:https://doi.org/10.1071/MF10302

Hobday, A. J., Okey, T. A., Poloczanska, E. S., Kunz, T. J., & Richardson, A. J. e. (2006). Impacts of climate change on Australian marine life: Part A. Executive

Summary. *Report to the Australian Greenhouse Office, Canberra, Australia. September 2006.*
Hobday, A. J., & Pecl, G. T. (2014). Identification of global marine hotspots: sentinels for change and vanguards for adaptation action. *Reviews in Fish Biology and Fisheries, 24*(2), 415-425. doi:10.1007/s11160-013-9326-6
Hobday, A. J., Smith, A. D. M., Stobutzki, I. C., Bulman, C., Daley, R., Dambacher, J. M., . . . Zhou, S. (2011). Ecological risk assessment for the effects of fishing. *Fisheries Research, 108*(2), 372-384. doi:https://doi.org/10.1016/j.fishres.2011.01.013
Hogg, O. T., Huvenne, V. A. I., Griffiths, H. J., Dorschel, B., & Linse, K. (2016). Landscape mapping at sub-Antarctic South Georgia provides a protocol for underpinning large-scale marine protected areas. *6*, 33163. doi:10.1038/srep33163 https://www.nature.com/articles/srep33163#supplementary-information
Holland, D. S., & Brazee, R. J. (1996). Marine Reserves for Fisheries Management. *Marine Resource Economics, 11*(3), 157-171.
Holland, D. S., & Ginter, J. J. C. (2001). Common property institutions in the Alaskan groundfish fisheries. *Marine Policy, 25*(1), 33-42. doi:http://dx.doi.org/10.1016/S0308-597X(00)00033-6
Hopf, Jess K., Jones, G. P., Williamson, D. H., & Connolly, S. R. (2016a). Fishery consequences of marine reserves: short-term pain for longer-term gain. *Ecological Applications, 26*(3), 818-829. doi:doi:10.1890/15-0348
Hopf, Jess K., Jones, Geoffrey P., Williamson, David H., & Connolly, Sean R. (2016b). Synergistic Effects of Marine Reserves and Harvest Controls on the Abundance and Catch Dynamics of a Coral Reef Fishery. *Current Biology, 26*(12), 1543-1548. doi:https://doi.org/10.1016/j.cub.2016.04.022
Horigue, V., Pressey, R. L., Mills, M., Brotánková, J., Cabral, R., & Andréfouët, S. (2015). Benefits and Challenges of Scaling Up Expansion of Marine Protected Area Networks in the Verde Island Passage, Central Philippines. *PLoS ONE, 10*(8), e0135789. doi:10.1371/journal.pone.0135789
Horta e Costa, B., Batista, M. I., Gonçalves, L., Erzini, K., Caselle, J. E., Cabral, H. N., & Gonçalves, E. J. (2013). Fishers' Behaviour in Response to the Implementation of a Marine Protected Area. *PLoS ONE, 8*(6), e65057. doi:10.1371/journal.pone.0065057
Hughes, T. P., Cameron, D. S., Chin, A., Connolly, S. R., Day, J. C., Jones, G. P., . . . Young, M. A. L. (2016). A critique of claims for negative impacts of Marine Protected Areas on fisheries. *Ecological Applications, 26*(2), 637-641. doi:doi:10.1890/15-0457
Islam, D., & Berkes, F. (2016). Can small-scale commercial and subsistence fisheries co-exist? Lessons from an indigenous community in northern Manitoba, Canada. *Maritime Studies, 15*(1), 1. doi:10.1186/s40152-016-0040-6

IUCN-WCPA. (2008). *Establishing Marine Protected Area Networks—Making It Happen.* . Paper presented at the IUCN World Commission on Protected Areas Washington, D.C.

Janmaat, J. A. (2011). Fishing in a Shallow Lake: Exploring a Classic Fishery Model in a Habitat with Shallow Lake Dynamics. *Environmental and Resource Economics, 51*(2), 215-239. doi:10.1007/s10640-011-9495-5

Jennings, S., & Kaiser, M. J. (1998). The Effects of Fishing on Marine Ecosystems. In A. J. S. J.H.S. Blaxter & P. A. Tyler (Eds.), *Advances in Marine Biology* (Vol. Volume 34, pp. 201-352): Academic Press.

Jennings, S., & Revill, A. S. (2007). The role of gear technologists in supporting an ecosystem approach to fisheries. *ICES Journal of Marine Science: Journal du Conseil, 64*(8), 1525-1534. doi:10.1093/icesjms/fsm104

Johnson, D. J., et al. (2017). *Social Wellbeing and the Values of Small-scale Fisheries*: Springer International Publishing.

Kahui, V., Armstrong, C. W., & Vondolia, G. K. (2016). Bioeconomic Analysis of Habitat-Fishery Connections: Fishing on Cold Water Coral Reefs. *Land Economics, 92*(2), 328-343. doi:10.3368/le.92.2.328

Kittinger, J. N., Finkbeiner, E. M., Ban, N. C., Broad, K., Carr, M. H., Cinner, J. E., . . . Crowder, L. B. (2013). Emerging frontiers in social-ecological systems research for sustainability of small-scale fisheries. *Current Opinion in Environmental Sustainability, 5*(3), 352-357. doi:https://doi.org/10.1016/j.cosust.2013.06.008

Kramer, D. L., & Chapman, M. R. (1999). Implications of fish home range size and relocation for marine reserve function. *Environmental Biology of Fishes, 55*(1), 65-79. doi:10.1023/a:1007481206399

Kronbak, L., & Lindroos, M. (2007). Sharing Rules and Stability in Coalition Games with Externalities. *Marine Resource Economics, 22*(2), 137-154.

Krueck., N. C., Ahmadia., G. N., Green., A., Jones., G. P., Possingham., H. P., Riginos., C., . . . Mumby., P. J. (2017). Incorporating larval dispersal into MPA design for both conservation and fisheries. *Ecological Applications, 27*(3), 925-941. doi:doi:10.1002/eap.1495

Kvamsdal, S., Eide, A., Ekerhovd, N.-A., Enberg, K., Gudmundsdottir, A., Hoel, A., . . . Sandal, L. (2016). Harvest control rules in modern fisheries management. . *Elem Sci Anthropocene, 4(1):000114*.

Larcombe, J., & McLoughlin, K. e. (2007). *Fishery Status Reports 2006: Status of Fish Stocks Managed by the Australian Government.* . Canberra: Bureau of Rural Sciences

Last, P. R., White, W. T., Gledhill, D. C., Hobday, A. J., Brown, R., Edgar, G. J., & Pecl, G. (2011). Long-term shifts in abundance and distribution of a temperate fish fauna: a response to climate change and fishing practices. *Global Ecology and Biogeography, 20*(1), 58-72. doi:doi:10.1111/j.1466-8238.2010.00575.x

Lauck, T., Clark, C. W., Mangel, M., & Munro, G. R. (1998). IMPLEMENTING THE PRECAUTIONARY PRINCIPLE IN FISHERIES MANAGEMENT THROUGH MARINE

RESERVES. *Ecological Applications, 8*(sp1), S72-S78. doi:10.1890/1051-0761(1998)8[S72:ITPPIF]2.0.CO;2
Lester, S. E., Halpern, B. S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B. I., Gaines, S. D., . . . Warner, R. R. (2009). Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series, 384*, 33-46. doi:10.3354/meps08029
Ling, S. D., Johnson, C. R., Ridgway, K., Hobday, A. J., & Haddon, M. (2009). Climate-driven range extension of a sea urchin: inferring future trends by analysis of recent population dynamics. *Global Change Biology, 15*(3), 719-731. doi:doi:10.1111/j.1365-2486.2008.01734.x
Little, L. R., Grafton, R. Q., Kompas, T., Smith, A. D. M., Punt, A. E., & Mapstone, B. D. (2011). Complementarity of No-Take Marine Reserves and Individual Transferable Catch Quotas for Managing the Line Fishery of the Great Barrier Reef Complementariedad de Reservas Marinas Sin Captura y Cuotas de Captura Individuales Transferibles para el Manejo de la Pesca con Anzuelo en la Gran Barrera Arrecifal. *Conservation Biology, 25*(2), 333-340. doi:10.1111/j.1523-1739.2010.01590.x
Little, L. R., Punt, A. E., Dichmont, C. M., Dowling, N., Smith, D. C., Fulton, E. A., . . . Gorton, R. J. (2016). Decision trade-offs for cost-constrained fisheries management. *ICES Journal of Marine Science, 73*(2), 494-502. doi:10.1093/icesjms/fsv206
Liu, O. R., Thomas, L. R., Clemence, M., Fujita, R., Kritzer, J. P., McDonald, G., & Szuwalski, C. (2016). An Evaluation of Harvest Control Methods for Fishery Management. *Reviews in Fisheries Science & Aquaculture, 24*(3), 244-263. doi:10.1080/23308249.2016.1161002
Lloret, J., & Planes, S. (2003). Condition, feeding and reproductive potential of white seabream Diplodus sargus as indicators of habitat quality and the effect of reserve protection in the northwestern Mediterranean. *Marine Ecology Progress Series, 248*, 197-208.
Lynham, J. (2014). How have catch shares been allocated? *Marine Policy, 44*(Supplement C), 42-48. doi:https://doi.org/10.1016/j.marpol.2013.08.007
Mangel, M. (2000). Trade-offs between fish habitat and fishing mortality and the role of reserves. *Bulletin of Marine Science, 66*(3), 663-674.
Mangi, S. C., & Roberts, C. M. (2006). Quantifying the environmental impacts of artisanal fishing gear on Kenya’s coral reef ecosystems. *Marine Pollution Bulletin, 52*(12), 1646-1660. doi:https://doi.org/10.1016/j.marpolbul.2006.06.006
Mardle, S., & Pascoe, S. (2002). Modelling the effects of trade-offs between long and short-term objectives in fisheries management. *J Environ Manage, 65*(1), 49-62.
Mascia, M. (2004). *Social Dimensions of Marine Reserves*.

Mason, J., Kosaka, R., Mamula, A., & Speir, C. (2012). Effort changes around a marine reserve: The case of the California Rockfish Conservation Area. *Marine Policy, 36*(5), 1054-1063. doi:https://doi.org/10.1016/j.marpol.2012.03.002

McCain, J. S. P., Rangeley, R. W., Schneider, D. C., & Lotze, H. K. (2016). Historical abundance of juvenile commercial fish in coastal habitats: Implications for fish habitat management in Canada. *Marine Policy, 73*, 235-243. doi:http://doi.org/10.1016/j.marpol.2016.08.009

McClanahan, T. R., & Kaunda-Arara, B. (1996). Fishery Recovery in a Coral-reef Marine Park and Its Effect on the Adjacent Fishery Recuperación de Pesquerías en un Parque Marino de Arrecife Coralino. *Conservation Biology, 10*(4), 1187-1199. doi:10.1046/j.1523-1739.1996.10041187.x

Moeller, H. V., & Neubert, M. G. (2012). Habitat damage, marine reserves, and the value of spatial management. *Ecological Applications, 23*(5), 959-971. doi:10.1890/12-0447.1

Moeller, H. V., & Neubert, M. G. (2015). ECONOMICALLY OPTIMAL MARINE RESERVES WITHOUT SPATIAL HETEROGENEITY IN A SIMPLE TWO-PATCH MODEL. *Natural Resource Modeling, 28*(3), 244-255. doi:10.1111/nrm.12066

Morgan, G. R. (1995). Optimal fisheries quota allocation under a transferable quota (TQ) management system. *Marine Policy, 19*(5), 379-390. doi:http://dx.doi.org/10.1016/0308-597X(95)00005-Q

Morison, A. K. (2004). Input and output controls in fisheries management: a plea for more consistency in terminology. *Fisheries Management and Ecology, 11*(6), 411-413. doi:10.1111/j.1365-2400.2004.00414.x

Munro, G. R. (1979). The Optimal Management of Transboundary Renewable Resources. *The Canadian Journal of Economics / Revue canadienne d'Economique, 12*(3), 355-376. doi:10.2307/134727

Murawski, S. A. (2010). Rebuilding depleted fish stocks: the good, the bad, and, mostly, the ugly. *ICES Journal of Marine Science: Journal du Conseil, 67*(9), 1830-1840. doi:10.1093/icesjms/fsq125

O'Leary, B. C., Winther - Janson, M., Bainbridge, J. M., Aitken, J., Hawkins, J. P., & Roberts, C. M. (2016). Effective Coverage Targets for Ocean Protection. *Conservation Letters, 9*(6), 398-404. doi:doi:10.1111/conl.12247

Ordoñez-Gauger, L., Richmond, L., Hackett, S., & Chen, C. (2018). It's a trust thing: Assessing fishermen's perceptions of the California North Coast marine protected area network. *Ocean & Coastal Management, 158*, 144-153. doi:https://doi.org/10.1016/j.ocecoaman.2018.03.034

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proc Natl Acad Sci U S A, 104*(39), 15181-15187. doi:10.1073/pnas.0702288104

Ostrom, E. (2008). The Challenge of Common-Pool Resources. *Environment: Science and Policy for Sustainable Development, 50*(4), 8-21. doi:10.3200/ENVT.50.4.8-21

Ostrom, E. (2009). A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science, 325*(5939), 419-422. doi:10.1126/science.1172133

Parks, D. o. N. (2013). South-east Commonwealth Marine Reserves Network management plan 2013-23. *Director of National Parks, Canberra.*

Pascoe, S., Doshi, A., Dell, Q., Tonks, M., & Kenyon, R. (2014). Economic value of recreational fishing in Moreton Bay and the potential impact of the marine park rezoning. *Tourism Management, 41*, 53-63. doi:http://dx.doi.org/10.1016/j.tourman.2013.08.015

Pascoe, S., Kahui, V., Hutton, T., & Dichmont, C. (2016). Experiences with the use of bioeconomic models in the management of Australian and New Zealand fisheries. *Fisheries Research, 183*(Supplement C), 539-548. doi:https://doi.org/10.1016/j.fishres.2016.01.008

Patterson, H., Noriega R, Georgeson, L., Larcombe, J., & Curtotti, R. (2017). Fishery status reports 2017. *Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra. CC BY 4.0.*

Pauly, D., Christensen, V., Guénette, S., Pitcher, T. J., Sumaila, U. R., Walters, C. J., . . . Zeller, D. (2002). Towards sustainability in world fisheries. *Nature, 418*, 689. doi:10.1038/nature01017

Pecl, G., Hobday, A. J., Frusher, S., Sauer, W. H. H., & Bates, A. E. (2014). Ocean warming hotspots provide early warning laboratories for climate change impacts. *Reviews in Fish Biology and Fisheries, 24*(2), 409-413. doi:10.1007/s11160-014-9355-9

Pecl, G., Ward, T., Doubleday, Z., Clarke, S., Day, J., Dixon, C., . . . Stoklosa, R. (2011). Risk Assessment of Impacts of Climate Change for Key Marine Species in South Eastern Australia. Part 1: Fisheries and Aquaculture Risk Assessment. *Fisheries Research and Development Corporation, Project 2009/070.*

Pelletier, D., García-Charton, J. A., Ferraris, J., David, G., Thébaud, O., Letourneur, Y., . . . Galzin, R. (2005). Designing indicators for assessing the effects of marine protected areas on coral reef ecosystems: A multidisciplinary standpoint. *Aquat. Living Resour., 18*(1), 15-33.

Pelletier, D., & Mahévas, S. (2005). Spatially explicit fisheries simulation models for policy evaluation. *Fish and Fisheries, 6*(4), 307-349. doi:10.1111/j.1467-2979.2005.00199.x

Péreau, J. C., Doyen, L., Little, L. R., & Thébaud, O. (2012). The triple bottom line: Meeting ecological, economic and social goals with individual transferable quotas. *Journal of Environmental Economics and Management, 63*(3), 419-434. doi:http://dx.doi.org/10.1016/j.jeem.2012.01.001

Pham, C. K., Diogo, H., Menezes, G., Porteiro, F., Braga-Henriques, A., Vandeperre, F., & Morato, T. (2014). Deep-water longline fishing has reduced impact on Vulnerable Marine Ecosystems. *Scientific Reports, 4*, 4837. doi:10.1038/srep04837

Pinhorn, A. T. (1988). Catchability in some of the major groundfish fisheries off the east coast of Canada. *Journal of Northwest Atlantic Fishery Science, 8*(15-23).

Pintassilgo, P., Kronbak, L. G., & Lindroos, M. (2015). International Fisheries Agreements: A Game Theoretical Approach. *Environmental and Resource Economics, 62*(4), 689-709. doi:10.1007/s10640-014-9850-4

Popova, E., Yool, A., Byfield, V., Cochrane, K., Coward, A. C., Salim, S. S., . . . Roberts, M. J. (2016). From global to regional and back again: common climate stressors of marine ecosystems relevant for adaptation across five ocean warming hotspots. *Global Change Biology, 22*(6), 2038-2053. doi:doi:10.1111/gcb.13247

Punt, A. E. (2006). THE FAO PRECAUTIONARY APPROACH AFTER ALMOST 10 YEARS: HAVE WE PROGRESSED TOWARDS IMPLEMENTING SIMULATION-TESTED FEEDBACK-CONTROL MANAGEMENT SYSTEMS FOR FISHERIES MANAGEMENT? *Natural Resource Modeling, 19*(4), 441-464. doi:10.1111/j.1939-7445.2006.tb00189.x

Punt, A. E. (2010). *Harvest Control Rules and Fishery Management* (Q. R. Grafton, R. Hilborn, D. Squires, M. Tait, & M. Williams Eds.). In: Handbook of Marine Fisheries Conservation and Management: Oxford University Press.

Punt, A. E., Smith, A. D. M., Smith, D. C., Tuck, G. N., & Klaer, N. L. (2014). Selecting relative abundance proxies for BMSY and BMEY. *ICES Journal of Marine Science, 71*(3), 469-483. doi:10.1093/icesjms/fst162

Rassweiler, A., Costello, C., Hilborn, R., & Siegel, D. A. (2014). Integrating scientific guidance into marine spatial planning. *Proceedings of the Royal Society B: Biological Sciences, 281*(1781). doi:10.1098/rspb.2013.2252

Rassweiler, A., Costello, C., & Siegel, D. A. (2012). Marine protected areas and the value of spatially optimized fishery management. *Proceedings of the National Academy of Sciences, 109*(29), 11884-11889. doi:10.1073/pnas.1116193109

Rayns, N. (2007). The Australian government's harvest strategy policy. *ICES Journal of Marine Science, 64*(4), 596-598. doi:10.1093/icesjms/fsm032

Reimer, M. N., & Haynie, A. C. (2018). Mechanisms matter for evaluating the economic impacts of marine reserves. *Journal of Environmental Economics and Management, 88*, 427-446. doi:https://doi.org/10.1016/j.jeem.2018.01.009

Reithe, S., Armstrong, C. W., & Flaaten, O. (2014). Marine protected areas in a welfare-based perspective. *Marine Policy, 49*, 29-36. doi:http://dx.doi.org/10.1016/j.marpol.2014.04.002

Ridgway, K. R. (2007). Long-term trend and decadal variability of the southward penetration of the East Australian Current. *Geophysical Research Letters, 34*(13). doi:doi:10.1029/2007GL030393

Rindorf, A., Dichmont, C. M., Levin, P. S., Mace, P., Pascoe, S., Prellezo, R., . . . Clausen, L. W. (2017). Food for thought: pretty good multispecies yield. *ICES Journal of Marine Science, 74*(2), 475-486. doi:10.1093/icesjms/fsw071

Roberts, C. M., Halpern., B., Palumbi., S., & Warner., R. (2001). Designing Marine Reserve Networks Why Small, Isolated Protected Areas Are Not Enough. *Conservation in Practice, 2*(3), 10-17. doi:doi:10.1111/j.1526-4629.2001.tb00012.x

Roberts, C. M., Hawkins, J. P., & Gell, F. R. (2005). The role of marine reserves in achieving sustainable fisheries. *Philos Trans R Soc Lond B Biol Sci, 360*(1453), 123-132. doi:10.1098/rstb.2004.1578

Roberts, C. M., O'Leary., B. C., McCauley., D. J., Cury., P. M., Duarte., C. M., Lubchenco., J., . . . Castilla., J. C. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proceedings of the National Academy of Sciences, 114*(24), 6167-6175. doi:10.1073/pnas.1701262114

Roberts, K. E., Valkan, R. S., & Cook, C. N. (2018). Measuring progress in marine protection: A new set of metrics to evaluate the strength of marine protected area networks. *Biological Conservation, 219*, 20-27. doi:https://doi.org/10.1016/j.biocon.2018.01.004

Russ, G. R., & Alcala, A. C. (2011). Enhanced biodiversity beyond marine reserve boundaries: The cup spillith over. *Ecological Applications, 21*(1), 241-250. doi:10.1890/09-1197.1

Russ, G. R., Alcala, A. C., Maypa, A. P., Calumpong, H. P., & White, A. T. (2004). MARINE RESERVE BENEFITS LOCAL FISHERIES. *Ecological Applications, 14*(2), 597-606. doi:10.1890/03-5076

Ryan, R. W., Holland, D. S., & Herrera, G. E. (2014). Ecosystem Externalities in Fisheries. *Marine Resource Economics, 29*(1), 39-53. doi:10.1086/676288

Sala, E., Aburto-Oropeza, O., Paredes, G., Parra, I., Barrera, J. C., & Dayton, P. K. (2002). A General Model for Designing Networks of Marine Reserves. *Science, 298*(5600), 1991-1993. doi:10.1126/science.1075284

Sala, E., Costello, C., Dougherty, D., Heal, G., Kelleher, K., Murray, J. H., . . . Sumaila, R. (2013). A General Business Model for Marine Reserves. *PLoS ONE, 8*(4), e58799. doi:10.1371/journal.pone.0058799

Sanchirico, J. N., & Wilen, J. E. (2001). A Bioeconomic Model of Marine Reserve Creation. *Journal of Environmental Economics and Management, 42*(3), 257-276. doi:http://dx.doi.org/10.1006/jeem.2000.1162

Savage, J. (2015). Australian fisheries and aquaculture statistics 2015. *Fisheries Research and Development Corporation project 2016-246. ABARES, Canberra, December. CC BY 3.0.*

Schaefer, M. B. (1957). Some Considerations of Population Dynamics and Economics in Relation to the Management of the Commercial Marine Fisheries. *Journal of the Fisheries Research Board of Canada, 14*(5), 669-681. doi:10.1139/f57-025

Sciberras, M., Jenkins, S. R., Mant, R., Kaiser, M. J., Hawkins, S. J., & Pullin, A. S. (2015). Evaluating the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries, 16*(1), 58-77. doi:doi:10.1111/faf.12044

Shephard, S., Brophy, D., & Reid, D. (2010). Can bottom trawling indirectly diminish carrying capacity in a marine ecosystem? *Marine Biology, 157*(11), 2375-2381. doi:10.1007/s00227-010-1502-9

Skirtun, M., Sahlqvist, P., & Vieira, S. (2013). Australian fisheries statistics 2012. *FRDC project 2010/208, ABARES, Canberra, November. CC BY 3.0.*

Smith, A. D. M., Smith, D. C., Haddon, M., Knuckey, I. A., Sainsbury, K. J., & Sloan, S. R. (2014). Implementing harvest strategies in Australia: 5 years on. *ICES Journal of Marine Science, 71*(2), 195-203. doi:10.1093/icesjms/fst158

Smith, A. D. M., Smith, D. C., Tuck, G. N., Klaer, N., Punt, A. E., Knuckey, I., . . . Little, L. R. (2008). Experience in implementing harvest strategies in Australia's south-eastern fisheries. *Fisheries Research, 94*(3), 373-379. doi:https://doi.org/10.1016/j.fishres.2008.06.006

Smith, J., & Metaxas, A. (2018). A decision tree that can address connectivity in the design of Marine Protected Area Networks (MPAn). *Marine Policy, 88*, 269-278. doi:https://doi.org/10.1016/j.marpol.2017.11.034

Smith, M. D., Lynham, J., Sanchirico, J. N., & Wilson, J. A. (2010). Political economy of marine reserves: Understanding the role of opportunity costs. *Proceedings of the National Academy of Sciences, 107*(43), 18300-18305. doi:10.1073/pnas.0907365107

Smith, M. D., & Wilen, J. E. (2003). Economic impacts of marine reserves: the importance of spatial behavior. *Journal of Environmental Economics and Management, 46*(2), 183-206. doi:http://dx.doi.org/10.1016/S0095-0696(03)00024-X

Spalding, M., Meliane, I., Milam, A., Fitzgerald, C., & Hale, L. (2013). *Protecting Marine Spaces: Global Targets and Changing Approaches* (Vol. 27).

Stefansson, G., & Rosenberg, A. A. (2005). Combining control measures for more effective management of fisheries under uncertainty: quotas, effort limitation and protected areas. *Philosophical Transactions of the Royal Society B: Biological Sciences, 360*(1453), 133-146. doi:10.1098/rstb.2004.1579

Stepanova, O. (2015). Conflict resolution in coastal resource management: Comparative analysis of case studies from four European countries. *Ocean & Coastal Management, 103*(Supplement C), 109-122. doi:https://doi.org/10.1016/j.ocecoaman.2014.10.017

Sundblad, G., & Bergström, U. (2014). Shoreline development and degradation of coastal fish reproduction habitats. *Ambio, 43*(8), 1020-1028. doi:10.1007/s13280-014-0522-y

Sutinen, J. G., & Johnston, R. J. (2003). Angling management organizations: integrating the recreational sector into fishery management. *Marine Policy, 27*(6), 471-487. doi:http://dx.doi.org/10.1016/S0308-597X(03)00079-4

Symes, D., & Crean, K. (1995). Privatisation of the commons: the introduction of individual transferable quotas in developed fisheries. *Geoforum, 26*(2), 175-185. doi:https://doi.org/10.1016/0016-7185(95)00024-F

Thrush, S. F., & Dayton, P. K. (2002). DISTURBANCE TO MARINE BENTHIC HABITATS BY TRAWLING AND DREDGING: Implications for Marine Biodiversity. *Annual Review of Ecology and Systematics, 33*(1), 449-473. doi:10.1146/annurev.ecolsys.33.010802.150515

Turner, S. J., Thrush, S. F., Hewitt, J. E., Cummings, V. J., & Funnell, G. (1999). Fishing impacts and the degradation or loss of habitat structure. *Fisheries*

Management and Ecology, 6(5), 401-420. doi:10.1046/j.1365-2400.1999.00167.x

Udumyan, N., Ami, D., & Cartigny, P. (2010). *Integrating habitat concerns into Gordon-Schaefer model*. Retrieved from https://halshs.archives-ouvertes.fr/halshs-00520328

UN. (2015). Transforming Our World: The 2030 Agenda for Global Action. *United Nations, New York*.

Upton, H. F., & Sutinen, J. G. (2005). *When Do Marine Protected Areas Pay? An Analysis of Stylized Fisheries.* Paper presented at the in AMERICAN FISHERIES SOCIETY SYMPOSIUM, 41; 745-758; Effects of fishing activities on benthic habitats: linking geology, biology, socioeconomics, and management; Benthic habitats and the effects of fishing by American Fisheries Society, Bethesda, Maryland.

Vieira, S. (2010). *Impact of the Structural Adjustment Package on the Profitability of Commonwealth Fisheries*: ABARES.

Ward, P., Marton, N., Moore, A., Patterson, H., Penney, A., Sahlqvist, P., . . . Woodhams, J. (2013). *Technical reviews for the Commonwealth Fisheries Harvest Strategy Policy: implementation issues*. Canberra.: ABARES, Report to client prepared for the Fisheries Research and Development Corporation.

Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature, 515*, 67. doi:10.1038/nature13947 https://www.nature.com/articles/nature13947#supplementary-information

Wells, S., Ray, G. C., Gjerde, K. M., White, A. T., Muthiga, N., Bezaury, C. J. E., . . . Reti, J. (2016). Building the future of MPAs – lessons from history. *Aquatic Conservation: Marine and Freshwater Ecosystems, 26*(S2), 101-125. doi:doi:10.1002/aqc.2680

Wetzel, C. R., & Punt, A. E. (2016). The impact of alternative rebuilding strategies to rebuild overfished stocks. *ICES Journal of Marine Science: Journal du Conseil*. doi:10.1093/icesjms/fsw073

White, C., Kendall, B. E., Gaines, S., Siegel, D. A., & Costello, C. (2008). Marine reserve effects on fishery profit. *Ecology Letters, 11*(4), 370-379. doi:10.1111/j.1461-0248.2007.01151.x

White, J. W., Scholz, A. J., Rassweiler, A., Steinback, C., Botsford, L. W., Kruse, S., . . . Edwards, C. A. (2013). A comparison of approaches used for economic analysis in marine protected area network planning in California. *Ocean & Coastal Management, 74*, 77-89. doi:https://doi.org/10.1016/j.ocecoaman.2012.06.006

Williams, I. D., Walsh, W. J., Claisse, J. T., Tissot, B. N., & Stamoulis, K. A. (2009). Impacts of a Hawaiian marine protected area network on the abundance and fishery sustainability of the yellow tang, Zebrasoma flavescens. *Biological Conservation, 142*(5), 1066-1073. doi:https://doi.org/10.1016/j.biocon.2008.12.029

Wilson, D., Curtotti, R., Begg, G., & Phillips, K. e. (2009). *Fishery status reports 2008: status of fish stocks and fisheries managed by the Australian Government*. Canberra: Bureau of Rural Sciences & Australian Bureau of Agricultural and Resource Economics.

Wing, C., Simon, K., & Bello-Gomez, R. A. (2018). Designing Difference in Difference Studies: Best Practices for Public Health Policy Research. *Annual Review of Public Health, 39*(1), 453-469. doi:10.1146/annurev-publhealth-040617-013507

Woodhams, J., Vieira, S., & Stobutzki, I. e. (2013). Fishery status reports 2012. *Australian Bureau of Agricultural and Resource Economics and Sciences, Canberra.*

Wooldridge, J. M. (2013). *Introductory Econometrics: A Modern Approach* (5th ed ed.). Mason, Ohio, USA: South-Western Cengage Learning.

Xuan, B. B., & Armstrong, C. W. (2017). Marine reserve creation and interactions between fisheries and capture-based aquaculture: A bio-economic model analysis. *Natural Resource Modeling, 30*(2), e12122-N/A. doi:10.1111/nrm.12122

Xuan, B. B., & Armstrong, C. W. (2018). Trading Off Tourism for Fisheries. *Environmental and Resource Economics*. doi:10.1007/s10640-018-0281-5

Yamazaki, S., Grafton, R. Q., & Kompas, T. (2010). Non-consumptive values and optimal marine reserve switching. *Ecological Economics, 69*(12), 2427-2434. doi:http://dx.doi.org/10.1016/j.ecolecon.2010.07.011

Yamazaki, S., Hoshino, E., & Resosudarmo, B. P. (2015). No - take marine reserves and illegal fishing under imperfect enforcement. *Australian Journal of Agricultural and Resource Economics, 59*(3), 334-354. doi:doi:10.1111/1467-8489.12078

Yamazaki, S., Jennings, S., Quentin Grafton, R., & Kompas, T. (2015). Are marine reserves and harvest control rules substitutes or complements for rebuilding fisheries? *Resource and Energy Economics, 40*(0), 1-18. doi:http://dx.doi.org/10.1016/j.reseneeco.2015.01.001